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Poelt J., 1969. *Bestimmungsschlüssel europäischer Flechten*. J. Cramer, Vaduz, 757 pp.

Trevisan R., Cappelletti E.M. & Caniato P., 1983. *Epicarp micro-characters in some european Angelica species*. Atti Accad. Naz. Lincei, Rend. Cl. Sci. fis., mat. e nat., 74: 77-82.

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RADIOCESIUM IN PLANTS OF FOREST ECOSYSTEMS

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Keywords: Forests, Radiocesium, Radioecology, Vegetation.

Abstract: This paper presents a review of the behaviour of radiocesium in plants of forest ecosystems, based on a screening of 375 articles. Particular stress is given to those factors which affect data variability in plants, such as vertical and horizontal patterns of radioactivity in soils due to interception, resuspension, wash-off, litter fall etc. The behaviour of radiocesium in different horizons of forest soils is discussed. The paper summarizes the main uptake mechanisms in fungi, lichens, bryophytes and higher plants, and the possible use of these organisms as bioaccumulators of radioactive deposition. For higher plants, the effects of several factors on root uptake are considered, such as pH, organic matter and clay content of different soil horizons, the concentrations of other ions in the soil solution, rooting depths, mycorrhiza, etc. Finally, the paper includes a discussion of translocation phenomena inside plants, of seasonal variation of radionuclide concentrations, and of the expression of radiocontamination of plant material. The expression of radiocesium concentrations on a water basis is suggested as being more appropriate than the usual expression on a dry weight basis for the solution of several radioecological problems.

Introduction

The first steps of radioecology date back to the late 1950's and early 1960's, when radioactive fallout from nuclear weapon tests in the atmosphere became a cause of growing concern to national authorities. At that time the main interest was in understanding the radioactive dose to man and its effects on human health, and most studies were focused on agroecosystems, a primary source of environmental radioactivity to the population.

The accident at the Chernobyl power station on April, 26, 1986 caused large quantities of radioactive substances to be emitted in the atmosphere. Radioactivity was transported and dispersed by air currents in the form of contaminated air plumes which reached several west European countries, mainly through wet deposition (e.g. Müller 1986, Persson *et al.* 1987, Duvernet 1989, Gudiksen *et al.* 1989). In the first year after the accident high radioactivity was measured in some agricultural products from western Europe, like vegetables and milk (e.g. Ward *et al.* 1989); this was mainly due to direct deposition. After one year, when root uptake became the dominant contamination pathway, agricultural products showed relatively low radioactivity, while higher values were often measured in products from natural and semi-natural

ecosystems, such as wildlife meat, mushrooms and berries. This led to a much more intensive study of natural ecosystems than in the early years of radioecology. The first studies already suggested that in forests and natural grasslands the cycling of radionuclides is much more difficult to understand than in simple agroecosystems. In particular, the great variability of radioactivity data in soils, plants and animals from natural ecosystems was, and still is, a serious handicap in the formulation of reliable predictive models.

Natural ecosystems differ from agroecosystems in several important features, the main one being that in agroecosystems soils are periodically ploughed and fertilized, while in natural systems they exhibit a more or less clear subdivision in an upper, mainly organic horizon and a lower, mineral horizon, differing in several important characteristics such as pH, moisture, nutrient status, biological activity etc. (Frissel *et al.* 1990). Biodiversity is also important, agroecosystems being often monocultures, natural ecosystems being, generally, species-rich. Forests, in particular, have a much more complicated structure than agroecosystems, and a much wider range of ecological conditions. A further, important and often neglected difference is

the extent of mycorrhization: in natural ecosystems most plants are in symbiosis with mycorrhizal fungi, a fact which very much complicates uptake and transfer mechanisms of radionuclides from soil to plants via roots.

The intensification of studies on natural ecosystems has had a profound influence on the development of radioecology. Many models have been developed thus far, and a great deal of information on parameter values was collected during decades of research. Nevertheless, environmental transfer parameter values, and, consequently, the results of models, still show a high degree of uncertainty, the reported values of some parameters ranging over one or two orders of magnitude or more (Miller & Hoffmann 1983, Monte 1990).

Whereas in early years rough, empirical parameters were often considered as adequate for estimating the dose to man, post-Chernobyl radioecology is characterized by an increasing interest in the fine mechanisms underlying the cycling of radionuclides in forests and grasslands, and in the causes of the high variability of the data. The numerous projects financed by national and international institutions forced scientists with very different backgrounds to collaborate in joint projects, which, due to the complexity of natural ecosystems, is fundamental for a sound development of ecology. Thus, radioecology is presently losing its character of an applied science depending on radiation protection, and is merging with ecology "tout court". The hitherto available results of radioecological investigations on natural ecosystems already provide a consistent body of knowledge, which may be of general interest for ecologists.

Radiocesium has received significant attention in radioecology because of its persistence in the environment, its prominence in nuclear fallout and the nuclear fuel cycle, and its significant contribution to external and internal radiation exposure to man.

Aim of this paper is to review the main results concerning radiocesium cycling in natural ecosystems, and especially in Boreal and Temperate forests of western and Central Europe. The main focus will be on long-distance dispersal wet deposition, as that originating from nuclear weapons tests and from the Chernobyl accident in western Europe (in the immediate vicinity of Chernobyl there was no rainfall during and after the accident). We shall try to especially stress those factors which are of general interest for plant ecology, and

especially those which affect data variability, as this is one of the major issues in radioecology, as it is in general ecology.

HORIZONTAL PATTERNS AND VERTICAL DISTRIBUTION OF RADIOACTIVITY IN SOILS

Although the behaviour of radiocesium in agricultural plants is relatively well-understood (see e.g. Coughtrey & Thorne 1983), before the Chernobyl accident much less was known about the mechanisms controlling radiocesium availability in plants and soils of natural ecosystems. In the radioecological literature, reference to high inter- and intraspecific variability in radioactivity data of forest or grassland plants from the same site is more the rule than the exception; large differences were also found among contiguous individuals of the same species (see e.g. Ernst & van Rooij 1987, Cooper & McHugh 1983, Killey *et al.* 1988, Horrill *et al.* 1990, Cooper & Mattie 1990, Nelin & Nylén 1994).

One of the reasons for such a situation is the very uneven patterning of radioactivity in undisturbed soils. For example, Niemann *et al.* (1989) found that in West Germany the deposition of radionuclides was extremely heterogeneous: within one kilometer, variations up to 30-fold occurred. Such a situation is more the rule than the exception, and is evident on different spatial scales. This may be due to differences in deposition (the quantity of radioactivity falling on a given surface during a given period), or to factors which determine an uneven repartition of radioactivity in soils (concentration factors).

There are three known pathways for radionuclide deposition from the atmosphere into the soil: a) gravitational precipitation of large particles with size exceeding 0.01 mm, b) washout of radioactive aerosols by atmospheric precipitation, c) dry fallout of small particles due to turbulent diffusion in the atmosphere (Tikhomirov & Shcheglov 1994). In the following we shall distinguish between direct and indirect deposition; the former derives from direct atmospheric fallout (primary direct deposition) or from resuspension phenomena (secondary direct deposition); the term "indirect deposition" refers to radioactivity deposited through the plant canopy by wash out, leaching, litter fall, etc. After deposition has occurred, patterns in soil radioactivity concentrations may arise, due to the action of biological or physical factors. Thus, in complex

systems differences in soil radioactivity do not necessarily reflect differences in deposition. This means that in such cases it might be difficult to reconstruct direct deposition patterns from measurements of radioactivity concentrations in soils. Incidentally, it should be underlined that in the literature it is not always clear whether differences in soil radioactivity are due to differential deposition, or to the effects of different concentration factors.

Radioactivity patterns in soils might be observed on different geographic scales: a) on a regional scale, b) on a local scale, c) at community level, d) within a given community. On the regional and local scales differences in rainfall are the most important factor creating radioactivity deposition patterns. Particularly in agroecosystems, and especially in monocultures of herbaceous plants, most of the radioactivity patterns observed in the soils are mainly due to differential deposition, and therefore are most evident on a regional or local scale. Below the local scale, a large range of soil activities and a high spatial variability between and within different plant communities were reported by several authors. Especially in structurally complex systems, as in multi-layered natural forests, it is important to distinguish between total deposition and radionuclide concentrations in soils. The latter depend on several factors besides deposition, such as: structure of the canopy, topography, soil features, nutrient cycling within the ecosystem, etc.

Both horizontal and vertical radioactivity patterns can be detected in the structured soils of natural ecosystems; these are of importance for understanding the high data spread of radioactivity data in plants, and for planning adequate sampling strategies in studies aiming at quantifying soil-to-plant transfer factors. Disregarding horizontal and vertical radioactivity patterns might result in such a high data variability that any generalization, not to speak of modelling, could be severely hampered.

We shall first discuss the main factors responsible for horizontal, then those for vertical patterning of radioactivity in the soils, giving particular emphasis to wet deposition, as this was the main deposition form for Chernobyl-derived radiocesium in western Europe.

Horizontal patterns

Direct primary deposition

Precipitation - On a regional or local scale the

most important factor creating horizontal radioactivity patterns is the amount of precipitation, followed by the type of rainfall. Of course, in the case of wet deposition, horizontal deposition patterns are directly related to patterns in precipitation. In general, forest stands occur on a much wider variety of topographical conditions than most agroecosystems. This might be a further reason for the higher variability of radioactivity data found in forests. Not only the quantity, but also the type of precipitation may be important. ApSimon *et al.* (1992) developed a dynamic model of the life-cycle of a convective storm, indicating that the type of wet precipitation may be one of the causes of the high variability in the deposition on a regional scale; rain originating from different phases of cloud formation is likely to scavenge radionuclides with different efficiencies, and deposit them at different rates.

Interception and retention by tree canopies -

Below the local scale, retention and interception are the most important pattern-producing phenomena. Interception by vegetation may retain large quantities of deposits. In particular, trees are efficient aerosols interceptors (Yamagata *et al.* 1969, Bunzl & Kracke 1988, Desmet & Myttenaere 1988, Sokolov *et al.* 1990), see Proehl & Hoffman (1994) for a review. It is assumed that the tree, understory and organic layers intercept some fraction of the deposited radionuclides, called "interception fraction". Intercepted radionuclides are then distributed among these compartments in proportion to the biomass of each compartment. Of course, the biomass of trees is the highest in forest ecosystems, and trees are among the most efficient interceptors of radioactive particles. The interception fraction is often estimated to be 0.2 to 0.25 for agroecosystems (Proehl & Hoffman, 1994), whereas that for forests it is much higher, around 0.7-0.8 (Ronneau *et al.* 1987, Bunzl *et al.* 1989, Schimmack *et al.* 1991). The studies carried out in the vicinity of the Kyshtym and Chernobyl radiation accidents in Russia and Ukraine (Kharitonov 1973), and during field model experiments (Tikhomirov & Shcheglov 1990), demonstrated that 60-90% of radionuclides falling on forests were initially intercepted by tree crowns. Tobler *et al.* (1988) found that the initial concentration of radiocesium depended on the relative active surface (cm^2/g) exposed to the fall out. If this is a general phenomenon, we should find higher concentrations in conifers than in broad-leaf trees, and higher concentrations in shrubs than in annuals (Nelin & Nylén 1994). Tree canopies, and

particularly trees with large leaf area, have a high collection efficiency for aerosols particles (Yamagata *et al.* 1969, Ronneau *et al.* 1987, Desmet & Myttenaere 1988, Sokolov *et al.* 1990, Rauret *et al.* 1994). Interception rates depend on leaf morphology; pubescent leaves are more effective than smooth leaves in collecting aerosols (Wedding *et al.* 1975). The structure of the forest canopy is also important in determining the extent of interception phenomena. The more dense the canopy, the highest will be the quantity of water retained by tree crowns. According to Fraiture (1992) the main tree species of Europe can be arranged in the following order of increasing canopy density, and hence of increasing retention capacities: *Larix*, *Pinus*, *Betula*, *Quercus* (deciduous species), *Carpinus*, *Fagus*, *Abies*, *Picea*. Retention values of radiocesium fallout by spruce crowns are generally high, ranging from 70% (Bunzl *et al.* 1989b), 80% (Ronneau *et al.* 1987), and even 90% (Melin & Wallberg 1991, Melin *et al.* 1994) of the total. The retention of radiocesium by spruce needles may be described as a double exponential curve (Sombre *et al.* 1990). According to Bergman *et al.* (1991) a large fraction (probably more than 50%) of radiocesium in the fallout over coniferous forests will be initially retained in the canopy under both wet and dry deposition, at least under the circumstances prevailing in the Chernobyl case. The season of fallout is also important; when it occurs in a deciduous forest in winter time, interception will be obviously small. Melin & Wallberg (1991) reported an interception of less than 35% of the dry deposited radionuclides in Swedish unfoliated deciduous beech, birch and alder stands. Feige *et al.* (1988) working in southern Germany, found that in 1986 the trees and shrubs which showed highest radioactivity values were those characterized by early leafing, all others being much less contaminated.

Horizontal patterns due to interception reflect the horizontal patterning of the intercepting structures. Thus, soil deposition will be highest in openings of the forest canopy, lowest under the canopy. Padovani *et al.* (1990), in NE Italy, found large differences in soil radiocesium between conifer forests and adjacent grasslands, with a significantly greater activity in the latter; on the contrary, there was no significant difference between grasslands and adjacent broad-leaved stands, which were unfoliated at deposition time. Similar results were obtained in Germany by Bunzl *et al.* (1989b) and in Belgium by Guillitte *et al.* (1990a); the latter authors observed a

wide variation of radiocesium activities in forest soils, with differences varying from +50% to -50% compared with the deposits in a clearing; deposits in a deciduous stand were close to values observed in clearings, while those in coniferous stands were considerably lower; furthermore, the highest deposits were found on the edge of stands, with an average increase of 18% compared with stands in clearings, and two times higher than those found within the stand. According to Fraiture (1992) in a natural forest the soils of large openings can receive 100% of total wet deposition, those under dense coniferous stands practically nothing. Raitio & Rantavaara (1994) claim that the spatial variation of soil radioactivity under and between tree crowns is probably greatest immediately after radioactive deposition has taken place; later on, differences tend to even out, and can sometimes even be reversed.

Similar considerations apply for dry deposition, with some important differences. Existing measurements of dry deposition of radioactive aerosols to forest canopies have usually been made as part of larger studies on radioactive aerosol deposition to different types of surface (Roed 1990) and are generally restricted to estimates of gross deposition velocities. Unfortunately, such studies reveal little about the detailed vertical and lateral distribution of deposits within canopies following interception (Shaw *et al.* 1994, Kinnersley *et al.* 1994). According to Bunzl & Kracke (1988) the average rate of total dry deposition over a long period was c. nine times greater in spruce forests than in adjacent grasslands. The filtration of a depositing aerosol in spruce canopies was studied experimentally by Shaw *et al.* (1994): filtration occurs rapidly at the upper and outer edges of an individual tree, resulting in a marked exponential attenuation of deposit within the canopy, from the topmost tissues downwards; horizontal differences in deposition become less as the canopy is penetrated vertically, contamination becoming uniformly distributed near the base of the canopy. Deposition to needles is significantly greater than that to stems, with a marked distinction between needle and stem contamination at the top of the canopy, lessening with vertical penetration of the canopy. Bunzl & Kracke (1988) and Bunzl *et al.* (1989b) observed differences of only 20 to 30% between the total radiocesium deposits on a spruce stand and neighbouring grasslands, with higher values in forests due to interception by tree canopies; according to Fraiture (1992) this is due to the fact that interception by tree crowns is much larger for

dry than for wet deposition. However, there is also evidence that in semi-arid ecosystems, such as Mediterranean forests in summer, dry deposition following resuspension could play a very important role in masking horizontal patterns, because of interception and retention by tree canopies; Rauret *et al.* (1994), studying an evergreen Mediterranean forest in Spain, did not observe clear differences between the amount of particles collected under and outside the tree canopy, and related this fact to the large increase in the total amount of collected particles during summer, due to resuspension phenomena.

Interception and retention by understory plants -

The more multi-layered a forest is, the more complicated will be the radioactivity pattern in the soils due to interception. The previous considerations concerning trees may be equally applied to tall and small shrubs. Interception and retention by cryptogams may be an important additional cause of sharp horizontal patterns in soil radioactivity as well. A thick carpet of lichens, and especially of mosses, is able to retain large quantities of water. Bryophytes, in particular, are often an important component of forest ecosystems, especially in humid and rainy areas, where their biomass is highest; they have a high capacity of absorbing liquid water, acting as intermediate sink for radiocesium. According to Giovani *et al.* (1994), 2 cm thick carpets of the moss *Ctenidium molluscum* can retain c. 12 l of water per square meter. Thus, when growing directly on soil, bryophytes can retain a great share of the total deposition, with a low transfer to soil. In ecosystems with an important bryophyte component, measurements of radioactivity in the soils (excluding the overlying bryophyte mats) do not provide a good estimate of total deposition, as most of the radioactivity is retained in the bryophyte mats. When bryophyte mats are patchy, a strong horizontal variation of radioactivity in forest soils can be expected, with higher values in areas not covered by bryophytes. Forest canopy, lichen and moss mats have a water saturation point, after which they are no more able to retain water (Giovani *et al.* 1994). Typically, trees may be exposed to c. 2-3 mm of rainfall before they lose their capacity to retain precipitation without extensive throughfall (Bergman pers. comm.); for many cryptogams the saturation power is much higher. The highest the saturation power, the highest will be the retention of deposition, and, consequently, the sharpest will be the horizontal

pattern between areas covered by vegetation and bare areas. Of course, the sharpness of the ground radioactivity pattern will be different if deposition occurs in an already wet or in a dry canopy.

Although forest canopies and moss carpets obviously have a higher interception and retention power, also grass mats are able to retain large quantities of radiocesium. Caput *et al.* (1990) studying permanent pastures, found that one year after the Chernobyl accident the activity of radiocesium in the vegetation was still relatively high, and independent of soil properties. This was explained by the persistence of radiocesium in the perennial plant bases which were exposed to the radioactive cloud in the period of the accident. Hereafter, the activity was observed to decline gradually, leading to transfer factors which are imputable to the usual soil-to-plant transfer mechanisms, and which are strongly related to soil properties. The importance of radionuclide retention in the basal parts of perennial grasses was already recognized in earlier studies on uptake of radioactive substances by plants exposed to weapon fall-out (Scott Russell 1966). This means that even in an apparently homogeneous grassland, different life-forms of herbaceous species might contribute to micropattern phenomena. These micropatterns, however, will be never so pronounced as those observed within forests; they could play an important role only in very open grasslands, with large patches of bare soil. Otherwise, in more or less homogeneous grasslands, the most important primary cause of soil radioactivity patterns is topography.

Direct secondary deposition (resuspension)

Not all of the deposited radioactivity is incorporated into the ecosystem. Following deposition unto the plants and soil, a continuous process of radionuclide removal begins. Aerosol-bound radionuclides may be shaken off from vegetation by wind action. Resuspension can be important for prolonging the availability of contaminants in the environment. Contaminated mineral soil particles can be resuspended back into the air by animal disturbance (Sumerling *et al.* 1984), wind (Anspaugh *et al.* 1975) or raindrop splash (Mazurak & Mosher 1968). This process, according to Miller & Hoffmann (1983), may lead to a rapid initial loss of material immediately after deposition. A review of resuspension data concerning the post-Chernobyl contamination in

Europe was published by Garland & Pattenden (1990). These authors concluded that resuspension from mineral soils may result in measurable activities in air for a period of at least 3 years following deposition. Green & Dodd (1988) found that, by dry weight, 7% of a herbage sample from a sheep paddock was actual soil; in terms of activity, however, that soil accounted for 74% of the total radiocesium activity.

Submicron particles can be generated from the surfaces of plants, especially during periods of rapid growth and high transpiration rates; radioactivity associated with these particles can be removed from vegetation by wind action (Beauford *et al.* 1975). Experimental results on spruce trees by Shaw *et al.* (1994) indicate that the bulk of dry aerosol deposits initially resides at the upper and outer edges of individual trees, which means that the potential for prompt resuspension by wind action following deposition is considerable. In the slightly longer term, however, the rate of field loss deposit in the actively growing upper and outer tissues will be greater than in older tissues which are less contaminated, and the problem of redistribution of deposit is therefore likely to be particularly acute immediately after the deposition event. The quantitative importance of wind removal in different ecosystems, in any case, remains to be evaluated. This factor might be of relevance only on a local level, and in special cases (see also Hinton 1991). The role of resuspension phenomena might have been overestimated, since most early studies were carried out in the dry areas of the southwestern United States, where resuspension is much more important for the contamination of plants and the dose to man than in humid temperate ecosystems, where soils have a well-developed upper organic layer. For example, studies based on resuspension sampling at several sites in Finland (Aaltonen *et al.* 1990) revealed that the primary source of the resuspended radiocesium was the local forest, i.e. that contributions associated with long-range transport of resuspended matter were too low to be significant. In Finnish forests the potential for contamination of the vegetation due to soil particles is relatively small; the roughly 3 to 5 times higher resuspension factors obtained at about the same time in Denmark (Aarkrog *et al.* 1988a,b), with a resuspension of about 3% during November 1986-October 1987, are typical of agricultural systems with considerable influence of wind erosion. In the semi-arid areas of Mediterranean Europe, however, resuspension could play a more important role; the

results obtained by Rauret *et al.* (1994) in a *Quercus ilex* forest of Spain, suggest that the Mediterranean climate, characterized by summer water stress and heavy rain storms in autumn, facilitates resuspension in summer and washing-off aerosols in autumn.

The factors influencing horizontal radioactivity patterns due to resuspension are the same as discussed for primary direct deposition.

Indirect deposition

Wash off - In structurally complex ecosystems only a minor part of deposition immediately reaches the ground, a considerable share being intercepted by vegetation. Parts of it are adsorbed by leaves, part are washed off by precipitation. Wash-off can occur as through-flow, when the water drips from the leaves, or as stem-flow. The radioactivity absorbed by plants will eventually reach the ground through leaching or litter fall. Thus, the structure of vegetation is another important cause of horizontal patterning of radioactivity. Patterning will be most pronounced in structurally complex systems, such as multi-layered forests, less pronounced in simpler systems, like grasslands.

If rainfall is sufficiently intense or long-lasting, the forest canopy reaches saturation, and water starts dropping to the forest floor. Wash-off removes the material from the surface of plants, leaching removes that incorporated inside the plants. Like direct wind removal, wash-off seems to be most effective immediately following deposition, while leaching may be a long-lasting phenomenon. According to Bunzl *et al.* (1989b) the half-life of radiocesium in spruce crowns was of 90 days in the first 130 days after deposition, and 230 days between the 131 and 600th day following deposition. Tobler *et al.* (1988), in Switzerland, estimate the half life of needles sprouted in 1985 to be 175 days, and 115 days for twig-wood within the period 50-240 days after the Chernobyl accident. From data on throughfalling rain and litterfall in a Norwegian spruce stand (Bergman *et al.* 1988) a corresponding half-life of 140 days was estimated for needles over the period 35-180 days at a boreal latitude. In a Swedish Scots pine forest half-lives of 114 days were estimated by Bergman *et al.* (1988) for the period 40-200 days, and 250 days for the period 200-360 days.

The branching geometry of trees is responsible for an uneven distribution of water washed down from the crown. Two main branching types can be distinguished: A) monopodial (centrifugal)

branching; typical of many conifers, this is characterized by the indefinite growth of the apical bud, from which lateral buds arise following the growth of the trees. In many forest species (e.g. *Picea*, *Abies*, *Taxus* etc.) lateral branches tend to grow at obtuse angles with respect to the trunk. The result is that rainfall (or snow) tends to fall to the ground from the margins of the crown. Hence, soil radioactivity is highest below the margins of the crown, lowest near the trunk (Guillitte *et al.* 1989, 1990a; Heinrich *et al.* 1989). Guillitte *et al.* (1990a) calculated that in conifers stemflow is less than 1% of rainfall for young stands, and about 3% for old stands. B) Sympodial (centripetal) branching; this, typical for deciduous trees, is characterized by the main apical bud being rapidly substituted by a secondary bud, which holds the dominance for a certain time, until it is substituted by another secondary bud. The result is a centripetal branching pattern, in which secondary branches tend to form acute angles with respect to the trunk. Hence, stemflow tends to concentrate from the crown to the trunk, and soil radioactivity concentrates around the trunk, being lowest below the crown margins (Heinrich *et al.* 1990, Schimmack *et al.* 1991). For deciduous trees the quantity of stemflow obviously depends on the season; Schnock (1967) carried out measurements for different trees in the leafed and leafless phases, and found considerable differences in stemflow, the highest values being reached during the leafed season. In old beech stands stemflow may amount to 6% of rainfall; this can explain the differences in soil radiocontamination found around the foot of deciduous trees, as brilliantly demonstrated by Guillitte *et al.* (1990a). In mixed forest this can result in a very complex pattern of horizontal deposition, which is further complicated by the topographical factors discussed above.

Litter fall and leaching are probably less important in pattern formation than physical factors. However, in mixed forests hosting both coniferous and deciduous trees horizontal radioactivity patterns in the soils will be enhanced in time, due to the higher efficiency of deciduous trees in nutrient cycling.

Leaching - The leaching of nutrients from the above-ground parts of plants due to the influence of acid rain and fog has been the subject of much study during the past few decades (Tukey 1980, Ivens *et al.* 1990). Leaching, although being a minor source of elements compared with litter fall, may be of significance in the balance of elements and in soil

micropattern formation within the forest. When radioactivity has been redistributed into the biologically active parts of the plant, rain operates as an external transfer agent. The leaching of radiocesium has been studied by several authors, such as Franklin *et al.* (1967), Waller & Olson (1967) and Nylén & Ericsson (1989). Even when rain has been eliminated by washout a fraction of radiocesium deposited onto the plant surfaces, leaching can be active for a long time. This means that transfer of radionuclides from leaves to soil via rain is effective not only at deposition time (Ronneau *et al.* 1991). Contrary to simple wash-off, leaching is more related to the physiological activity of the plant. According to Sombre *et al.* (1990), in a spruce forest, in first time radiocesium is preferentially retained by spruce needles and then washed off gradually or incorporated in the leaves; in the long term, the incorporated cesium is leaked out from the needles, and more cesium is collected under the canopy than in clearings, giving rise to horizontal radioactivity patterns in the soils. Ronneau *et al.* (1991) suggest that after a mid-term steady state has been attained in trees, radioactivity is being leached at a rate determined by their physiological activity; leaching may occur throughout a growing season, but it is the greatest just before maturity and death of the foliage (Long *et al.* 1956). Rain falling as a light continuous drizzle seems to be more efficient as a leaching agent than is a large quantity of rain falling during a short period (Miller & Hoffmann 1983). Similarly, the leaching occurring at the beginning of a rain period is higher than at the end of the period (Witherspoon 1962, 1964). Leaching may be also enhanced by high temperatures (Tukey 1980).

Litter fall - In a mixed forest, different species of trees may have absorbed different shares of the total deposit. Accordingly, litter fall will produce a differential enrichment in radionuclides of the respective litter layers. This may also apply to different leaves of the same tree: Vallejo *et al.* (1990) demonstrated that 3-year-old *Pinus* needles still contained relatively high activities of radiocesium, in contrast with the much lower values of younger leaves, which were not directly exposed to direct deposition. This may cause yearly differences in radiocesium output from leaves to the ground via leaf fall. In conifers, washout takes place very quickly after direct deposition; the amount remaining in the needles is deposited gradually when these fall, thus spreading the initial residual deposit

over several years; as the average life of a needle is of 3-6 years, there will be a differential input of radiocesium to the ground via leaf-fall, with a maximum value corresponding to the fall of leaves which absorbed direct contamination (Guillitte *et al.* 1990a). The decomposition of coniferous needles is rather slow, resulting in a prolonged input period (Henrich *et al.* 1990). According to Bunzl *et al.* (1989b) c. 7% of the total radiocesium pool present in the canopy in a Norwegian spruce forest is lost by litter fall during the first year. Similar levels were found in Scots pine forests by Bergman *et al.* (1991). The contamination of deciduous leaves is high only in the first year after deposition, and their rotting is rather quick (one to three years, Henrich *et al.* 1990), which leads to a rather peak-shaped input term. Similar results were obtained by Nelin & Nylén (1994) in a Swedish Boreal forest.

Concentration factors

On a local scale, horizontal patterns due to concentration factors depend both on physical and biological phenomena. Among the physical phenomena, the most important is horizontal water circulation within the ecosystems, which is influenced by microtopography.

Topographical factors - Contrary to most agroecosystems, natural stands are often characterized by a more or less pronounced topographical patterning of the ground; the importance of topographical factors is especially evident in mountain areas. After deposition, radiocesium may penetrate into the soil, and become eventually adsorbed by organic colloids and clay within the soil, or be removed by run-off (Wise 1980). The component removed by run-off can be termed "direct run off" (Queirazza *et al.* 1988) or the "fast hydrological component" (Smith *et al.* 1987), as compared with the "slow erosional component" or "delayed removal" due to later transport of particulate by erosive processes (see e.g. Bonnett 1990).

Water accumulates in small depressions and along drainage channels, and, consequently, radioactivity is higher there than in raised portions of the ground (Henrich *et al.* 1990). This pattern is influenced by two further factors: a) intensity of rainfall, b) soil permeability; the latter depends mainly on soil texture, but also on climatic factors (e.g. freezing, previous precipitation). A difference of an order of magnitude in the radioactivity

deposition in mountain soils of the same type covered by the same amount of grass taken at a distance of a few meters was observed by Maubert *et al.* (1990). These authors found that in an Alpine environment runoff is responsible for the distribution of radionuclides, with a preferential fixation at the bottom of slopes. The intensity of rainfall is also important, as more intense precipitations will result in more intense runoff. Furthermore, the more impermeable are the upper soil horizons, the stronger is the horizontal patterning of radioactivity, because impermeable soils tend to favour runoff phenomena. Horrill *et al.* (1990) reported higher radiocesium activities on damp peats, bog/flushes and peaty grasslands, and lower activities on adjacent, drier, more mineral soils. It seems that, at least in Boreal forest ecosystems, runoff phenomena are of importance especially just after deposition; according to Nylén & Grip (1989) c. 7% of the initial fallout from a catchment in the Middle Boreal zone of Sweden was transferred by runoff during 1986; the main part was lost already within the four weeks after the Chernobyl accident. During subsequent years this leakage decreased to c. 0.2% per year. The fraction lost during 1987-1990 from peat, which constitutes the main source for loss of radiocesium from catchments in the Boreal zone, was of c. 1-2%.

A special case is that of deposition occurring on snow-covered areas. Gaare (1987b) found that on wind-exposed, and hence snow-free areas of Norway the activity of Chernobyl-derived cesium in vegetation was c. three times higher than on adjacent, snow-covered sites. The higher activity found in lichens from relatively dry snow-free sites was ascribed to an increased absorption potential of dry lichens (Gaare 1987b).

Another source of radioactivity concentration in forests is the differential accumulation of litter by wind action; this phenomenon has never been studied in detail, but might be of some importance in special cases.

Biological factors - The activity of living organisms can produce horizontal displacements of radiocesium in the soils. The effect of biological factors is felt only some time after deposition, and becomes increasingly important in time, considerably complicating the horizontal patterns of radioactivity. Three main groups of organisms are responsible for secondary horizontal patterns; in order of increasing importance, they are: fungi, higher plants, and animals.

Fungi - Contrary to those of most agroecosystems, the soils of natural ecosystems host an intense fungal life. Fungi not only act, with bacteria, as the main group of decomposers (saprophytic fungi), but are also important as parasites of living plants, and especially as symbionts with their roots (mycorrhiza). The mycelium of a single fungal individual can extend for considerable distances in the forest soil. Many fungi have a high affinity for radiocesium, which is absorbed by the mycelium and concentrated in the fruitbodies. As fungal structures in the soil are often perennial, and as movement of nutrients occurs through the hyphae, fungi may form a major pool of radiocesium in the soil. The dominant fungal populations in natural forests are able to immobilize and retain nutrients and, because of the shortage of mineral nutrients relative to available carbon, they may translocate ions within their hyphal biomass from dying and lysing to actively growing hyphal zones (Dighton & Boddy 1989). Fungal turnover rates and the degree of conservation of any element will regulate its availability to other components of the ecosystem (Oujia & Myttenaere 1994). Horizontal displacement of radiocesium by fungi from the firstly contaminated vegetative hyphae can occur through the following pathways: a) to the actively growing new mycelium, b) to fruitbodies, c) to the roots of symbiotic plants. Thus, fungi are probably the most important biological agent of horizontal and vertical displacements of radiocesium in the soil. Despite the fact that little attention has been devoted to micropatterns in soil radioactivity due to fungi, the few available observations suggest that fungi are able to concentrate large quantities of radionuclides at a very small spatial scale (corresponding to the size of the carpophores), at the same time depleting the radiocesium pool in the surrounding areas. Seeger & Schweinhaut (1981) measured radioactivity in fungal fruitbodies and in the surrounding soil, and found that in cesium-rich fungi a marked concentration had occurred over the cesium content of the soil. Bergman (pers. comm.) found considerable enrichment in radioactivity in small areas where fungi developed carpophores. According to Fraiture (1992) the fructifications of *Xerocomus badius*, which can weight up to 200 g and have concentrations of c. 10.000 Bq/kg fresh weight in the most contaminated regions of Europe, can contain up to 2000 Bq of radiocesium; as these fungi rot on an area of 1 to 2 dm², the activity is significantly enhanced on a microscale. Thus, in a mushroom-rich environment the primary pattern of

soil radioactivity will be considerably complicated by fungal action, and this phenomenon will tend to increase in time.

Higher plants - Horizontal displacements of radionuclides by higher plants are, of course, larger in forests than in grasslands. From the soil, trees and forest undergrowth accumulate radiocesium in proportion to the biomass of the various tree components or undergrowth. According to Van Voris *et al.* (1990) the total activity in a forest understory is very small in comparison to that in the tree canopy, as reflected in the comparative biomass of each. In grasslands, grasses are able to intercept large quantities of radiocesium, part of which is absorbed into the plant body, while part is transferred to the organic layer of the soil. A rough estimate of biological retention in forest plants in relation to the initial deposit was provided by Merish (1990): 5-10% in tree trunks and branches, 10-20% in root systems of trees and herbaceous plants, less than 1% in tree leaves, less than 1% in the aerial part of herbaceous plants. It is difficult to estimate the extent of horizontal displacements of radioactivity brought about by trees through root uptake and subsequent release from the canopy. In general, the root systems of trees do not extend very much beyond the limits of the crowns. A further complication is given by mycorrhizal fungi, which are supposed to extend widely in the forest soils, and which are a major source of elements for trees. Furthermore, many common and widespread understory plants, such as *Vaccinium myrtillus* or *Lycopodium* species, tend to spread vegetatively by means of rhizomes, which in some cases, as in *Vaccinium*, may form a dense network, which makes difficult the distinction of single individual plants. In certain beech forests of the southern Alps a single plant of *Vaccinium* can spread over a distance of tens of meters. Of course, the calculation of transfer factors from soil to plant, in such cases, becomes extremely difficult, especially if there is a sharp micropattern of radioactivity in the forest soils.

Animals - Horizontal displacements of radioactivity in forest soils due to animals are mainly related to herbivory. The first studies on the elimination of radiocesium by herbivores were carried out by Baker *et al.* (1970), and Baker & Dunaway (1975) on populations of small rodents. According to Bergman *et al.* (1994), herbivory is estimated to remove about 10% of annual net primary productivity on average, while 90% is

removed by decomposers. These figures, of course, are very variable according to the type of ecosystem: in some situations plants lose less than 2% to herbivores, whereas in exceptional cases the loss exceeds the whole annual production. Bergman *et al.* (1994) have tried to quantify the effect of herbivory by bank vole and moose in a Swedish Boreal forest. The results indicate that herbivory appears to be within an order of magnitude of litterfall. Transfer of radioactive cesium by herbivory is seen to be even higher in comparison to that by litterfall, also considering that old leaves of conifers tend to translocate radiocesium to other parts of the plant before falling off. There are several studies reporting particularly high radiocontamination in the meat of some herbivores which are supposed to feed, at least partially, on fungi (e.g. see Palo *et al.* 1991, Pastor & Naiman 1992). The excrements of such animals are an important source of radiocesium displacements within the forest. Furthermore, the availability of radiocesium from animal excrements is expected to be high and fast, which further emphasizes the relative importance of herbivory in the feed-back from vegetation to soil (Bergman *et al.* 1994). As many wild herbivores do not move randomly through the forest, a differential enrichment in soil radiocesium could be expected.

Vertical distribution

After each deposition event, most of the radioactivity that falls out to the ground is deposited in the uppermost soil layers. In endopercolative soils, as those prevailing in the Temperate and Boreal zones, soils tend to be leached, and radiocesium tends to migrate into deeper soil horizons. The knowledge concerning the transportation of radionuclides downwards through the soil is based mainly on laboratory experiments (Van Genuchten & Cleary 1979, Relyea *et al.* 1980), and also on research carried out in field conditions after the Chernobyl accident (Bunzl *et al.* 1989a). The vertical transport of cesium ions may result from: a) transport of ions by means of percolating water or by ion diffusion in the soil water, b) transport of ions bound to colloids, c) mechanical mixing caused by soil animals, d) redistribution by uptake and translocation in soil, as well as leakage and production of litter from plants and fungi. Different soil horizons have different biological, chemical and physical characteristics, which influence the degree of retention and migration of radiocesium through the profile. Thus, in most natural soils there is a clear vertical gradient

of radiocesium distribution. In a natural ecosystem, many plant species coexist, and concur with each other for space, light, water and nutrients. Since plants tend to avoid direct concurrence by developing their root systems in different soil horizons, another possible cause of the high variability of plant radioactivity data could be the different root depths, and the unequal distribution of radiocesium in the soil profiles. Vertical radioactivity patterns are at least as large as horizontal patterns, and also in this case it is important to acquire previous knowledge on the extent of vertical patterns and on the depths of root systems for calculating reliable transfer factors soil-to-plant.

The vertical distribution of radiocesium in soils is a function, besides of time from deposition, of the following main factors: A) Physical factors: 1) quantity and intensity of precipitation fallen since deposition, 2) physico-chemical properties of soils. B) Biological factors: 1) absorption and re-deposition by plants, 2) biological characteristics of soils.

Physical factors

Precipitation - In endopercolative soils, the vertical migration of radionuclides into the soil profile is determined in first line by the amount and intensity of precipitation, which may already influence the downward migration of cesium at deposition time. Different viewpoints were proposed about the role of intrasoil flow in the process of distribution of radionuclides in the soil. A "retardation factor" was suggested by Bachhuber *et al.* (1982) as one of the main parameters for estimating radionuclide migration in the soil profile. This is represented by the equation $R = W/N$, where W is the average velocity migration of capillary water in the soil, and N the velocity migration of the radionuclide. Experimental results obtained in the Chernobyl area in 1987-1988 by Prister *et al.* (1990) showed that more than 15% of the radiocesium backed from the 0-10 cm layer by intrasoil flow during this period. Tikhomirov *et al.* (1990b) suggested that radionuclides mainly migrate by diffusion and by mechanical intermixing of the upper soil layers by mesofauna. Data by Haugen & Uhlen (1992) from Norway demonstrate that heavy rainfall short after deposition causes a rapid penetration of 40% or more of the activity to a depth below the upper 4 cm. Similar observations were made in southern Germany by Block & Pimpl (1990)

and Bunzl *et al.* (1989a). According to Block & Pimpl (1990) the heavy precipitation at the peak of the airborne Chernobyl activity could have resulted in part of the cesium passing through the main humus cover rather quickly, before an equilibrium had established itself, and not being immobilized until it had reached the mineral soil. Similar effects were observed by Rühm *et al.* (1995), who investigated a coniferous forest of southern Germany. In 1986 they found 22% of the initial radiocesium due to Chernobyl in the L-horizon, 39% in the Of-horizon, 35% in the Oh-horizon and 4% in the Ah-horizon. Since heavy rainshowers were reported in this region during the period immediately following the accident, the initial deposition profile was explained by penetrating rain water transporting part of the initial fallout to deeper layers immediately after deposition. Such a phenomenon could be demonstrated by Schimmack *et al.* (1994), who found up to 30% radiocesium in the mineral layers of undisturbed forest soil cores after artificial precipitation of up to 30 mm/h, simulating a heavy rain shower.

Belli *et al.* (1990b) studied the radiocesium residence half-times in soils of different ecosystems in the Carnic Alps (NE Italy). In the top layers of soils from a beech forest the cesium residence half time was estimated at c. 10 years. High rainfall, measured in 1987 and 1988, is cited as one of the environmental factors which can explain lower cesium residence half-times found in certain types of horizons. Data from the western U.S. attest that field areas subject to moderate to heavy throughput of water will tend to have a less superficial distribution of radiocesium in the soil profiles, contrary to unwatered field areas (Miller *et al.* 1990). Schell & Tobin (1990) studied the vertical migration of radiocesium in the soils in ombrotrophic bogs: they found that climatic conditions dominate the transport of radiocesium in the unsaturated zone by infiltration caused by precipitation, and by capillary action caused by evapotranspiration. During wet years, the transport by infiltration could be great. If the same site was measured later, the vertical distribution of radiocesium would reflect the precipitation and evaporation cycles until the radionuclide reached the water table. Belli *et al.* (1994), studying a mixed forest in the eastern Alps, found detectable concentrations of Cs-134 down to 15 cm depth. The content of Cs-134 in the deeper layers of the soil was already observed just after the Chernobyl accident (Belli *et al.* 1990b). This apparent fast migration is attributable to the percolation of contaminated

rainwater falling during the passage of the Chernobyl cloud. Of course, the more intense and long-lasting are the precipitations during fallout, the deeper will be the initial penetration of the radionuclides through the soil profiles.

The long-lasting persistence of radionuclides in the organic soil horizons (see later), however, contrasts with many of the previous considerations. It seems that capillary water flow is important for vertical migration of radiocesium only immediately after deposition, whereas later the radiocesium is somehow fixed in the organic layers, and hence much less mobile. According to Bergman (pers. comm.), the vertical distribution established early in soil is very rapid, being affected by quantity and intensity of rainfall, as indicated by the relatively deep penetration directly after wet deposition. A study in the Chernobyl area (Kliashtorin *et al.* 1994) demonstrated that only a small portion of the radionuclides could migrate in the soil profile with the water flow; in particular, the washout of radionuclides with intrasoil vertical water flow from the forest litter ranged from 0.1 to 3.5% of the total amount of radionuclides, most of the radionuclides being absorbed from the water flow in the 5 to 10 cm soil layer. The A horizon, therefore, appears to be the main barrier for the radionuclides moving by water flow down to the boundaries of the soil profile and to underground waters.

In complex forests, through-flow and stemflow already produce an horizontal redistribution of precipitation. It must be added that the chemical properties of water coming from these pathways are different than those of rain water. These differences may have an influence on the leaching rates in forest soil, which could result in differential redistribution of radiocesium in the soil profiles at a very small scale. This phenomenon, however, has never been studied in detail.

Physico-chemical properties of soils - At equal amounts and intensity of precipitation, different soils may present widely different vertical patterns of radiocesium distribution, as these also depend on physical characteristics of the soil, such as permeability, texture, porosity etc, and by chemical features. The fate and behaviour of radiocesium in natural ecosystems, with respect to both transport through the soil profile and its availability to plants, is a function of its solution chemistry and physico-chemical interactions with the solid phase minerals and the organic matter in the soil (Pritchett 1979). Radiocesium movement and transfer in the food

chains is generally a function of the availability of both the element and its biological analogue, potassium.

Organic horizons - Forest soils are generally characterized by well-defined layers in which three superficial horizons very rich in organic matter can be distinguished. The L (litter) horizon consists of intact litter with little visible signs of decomposition; the F (fermentation) horizon, below the L horizon, consists of fragmented litter, and the H (humus) layer between the F layer and the mineral soil containing little or no mineral matter. When radiocesium reaches the ground, it generally accumulates in the litter fraction of the soil organic material. If the deposition is due to contaminated leaves, the litter will have to undergo decomposition (biodegradation) before cesium is made available for root uptake. The speed of decomposition depends on several factors, the main one being temperature: therefore, decomposition is slower in northern forest ecosystems, faster in Mediterranean ecosystems. Sauras *et al.* (1994), working in a Mediterranean evergreen forest, demonstrated that radiocesium present in litter presented an initial rapid leaching period, which corresponded to the fraction extracted with water from the initially contaminated leaves. Further radiocesium release was related to litter decomposition, and after three months around 70% of the initial radiocesium was transferred to the underlying layers, mainly the F layer. The decomposition speed of boreal coniferous forest litter is much slower, and extends over several years (Melin *et al.* 1994).

Organic layers contain complexing organic chemicals resulting from microbial decomposition of the litter and the humic and fulvic acids present. It is possible to distinguish a hologenic (O) and a hemiorganic (Ah) horizon, that generally have different characteristics. The cation exchange capacity is significantly higher in the hologenic horizons, and seems to depend mainly on the degree of fractioning. On the other hand, texture appears to be the main source of variation in the hemiorganic horizons (Andolina & Guillitte 1990). The type of humus prevailing in organic horizons differs among forest ecosystems: whereas under Boreal forests the Moder type of humus is prevailing, in Mediterranean forests the Mull-type humus is more frequent. Furthermore, the quantity of humus tends to decrease from northern forests on acid ground to Mediterranean forests on calcareous parent material, due to the increasing speed of decomposition

processes favoured by higher temperatures. Petersen & Luxton (1982) suggested an inverse relationship between weight of organic material in the upper soil layers and the total soil fauna biomass. Therefore, the Mull-like humus forms widespread in Mediterranean forests have a relatively high soil fauna activity (Sauras *et al.* 1994). The effect of humus type on the vertical distribution of radionuclides was studied by Andolina & Guillitte (1990); two main factors influence the migration rates: the first factor, which includes two components, i.e. the thickness and the particle size distribution of the hologenic horizons, affects the interception rate (see also Guillitte *et al.* 1989). The second factor is the decomposition rate in these horizons. Thus, two years after the Chernobyl fallout, very low quantities of deposits in the hologenic layers were detected in the Mull type of humus, whereas Moders, and in particular Dysmoders still contained up to 75% of the initial deposits. According to Guillitte *et al.* (1990c) the variation of deposits in hemiorganic horizons is inversely proportionate to the average thickness of hologenic horizons, and is less important in Moder type than in Mull type of humus. In natural forests, different humus types may occur within the same community, which might result in different vertical profiles of radiocesium distribution in soil at a very small scale. Rates and patterns of radiocesium release from different plant litters were studied by Clint *et al.* (1990); their results indicate that there are significant variations among species: *Calluna vulgaris* litter had the lowest rate of release of both radiocesium and potassium, and the lowest levels of microbial activity. According to Andolina & Guillitte (1990) an important fraction of radiocesium is bound to lignins in the hologenic horizons; the fraction bound to fulvic and humic acids closely matches availability measurements, and could be used as an indicator for availability.

Below the litter layer there is usually an organic layer consisting of more or less decomposed organic matter. Part of the radiocesium will be retained in the organic layers by negatively labelled sites in the organic matter. As generally the upper soil layers are poor in clays which can fix the cesium, this is mostly in a form which is ready available for biological uptake, and is likely to be absorbed into the plant through the root system (Boikat *et al.* 1978, Horak 1986).

However, roots, fungi, soil microflora and - fauna are included in the measurements of the vertical distribution of cesium in soils; thus, it

cannot be excluded that a share of the cesium measured in the soil is not readily available for uptake, being retained in the soil biomass (see later). Miller *et al.* (1990) reported that for forest areas of the eastern U.S., the fraction of the total inventory residing in the top 5 cm ranged from 30% to 85% and on average was 58%, while for the field areas the range was 10% to 59% with an average of 31%. This suggests that varying retention mechanisms are likely affecting the profiles. The vertical distribution of radiocesium in Podzol soils of a coniferous forest in Germany was studied by Römmelt *et al.* (1990), who found slow migration rates, especially from the organic horizons into the mineral soil. Also, Livens *et al.* (1991) studying upland areas soils throughout Europe, found that most of them contained illite or micaceous materials in the deeper horizons, which, in principle, can take up radiocesium; however, extensive sorption to clays does not occur, probably because of recycling of radiocesium in the organic horizons, which prevents it reaching the clay-rich deeper layers (see also Valcke & Cremers 1994). Belli *et al.* (1990b) studied the vertical migration of radiocesium in different ecosystems of the Carnic Alps (NE Italy): they found that radiocesium behaves differently in natural grasslands and in woodland soils; in the latter, the thick organic layers have a much stronger retention of radionuclides. The retention of radiocesium has been found to be more efficient in surface layers with a high K and humus content than in deeper pure mineral layers by Thiry *et al.* (1991). Similar observations were made early in the history of fallout investigations (Alexander 1967) and have more recently been reported by so many authors that the slow downward migration of radiocesium in organic horizons of forest soils can be considered as a well-established fact, which still awaits a clear causal explanation (see e.g. Adriano *et al.* 1981, Kühn *et al.* 1984, Melin & Wallberg 1991, Lindner *et al.* 1994, Belli *et al.* 1994, Fawaris & Johanson 1994, Melin *et al.* 1994, Strandberg 1994, Tikhomirov & Shcheglov 1994, Wirth *et al.* 1994). The thickness and nature of the organic horizon were found to be primary parameters which greatly influence radiocesium migration in a forest soil profile. Thiry & Myttenaere (1993), however, mention that in spite of its very high cation exchange capacity, organic matter alone is not a direct source of cesium fixation; a higher percentage of organic matter has not been related to higher cesium retention in the different horizons. According to De Preter (1990), more important is the presence of low levels of micaceous minerals, providing highly

specific sites for cesium fixation. Humic compounds and potassium, which abound in organic-rich horizons would have, however, an indirect but important effect on cesium fixation. Another possible explanation is the phenomenon whereby the forest litter layer acts as a buffer by absorbing water from precipitation, thus slowing the movement of trace constituents into and through the mineral soil layers (Adriano *et al.* 1981). A further, perhaps more plausible hypothesis is that the main cause is biological retention by fungi and other microorganisms whose biomass is highest in the organic layers. This point will be treated in more detail further on.

Mineral horizons - Below the organic layers, many mature forest soils have mineral layers, whose chemico-physical characters can be widely different: some soils, like Podzols developing under Boreal forests, have a mineral layer with a sandy texture, whereas in many brown soils of the Temperate zone the texture is mainly clay. The relative amounts of sand and clay in mineral horizons influence the migration speed of radiocesium through the soil profile. In sandy soils migration is faster, due to more rapid water penetration into the ground. Evidence of deeper penetration in sandy as opposed to finer-textured soils has also been found in the course of studies on erosion processes (Menzel *et al.* 1987). High clay content generally implies a slow migration of radiocesium, both because of the lower infiltration speed in clay layers, and of their higher exchange capacity. The ion-exchange reaction depends on the clay content of the soils, and on the concentration of other exchangeable ions, especially potassium, in the soil solution (Bange & Oversweet 1960). The adsorptive capacity of clays with regard to cesium has been known for several decades (Squire & Middleton 1966, Witherspoon 1964). Illite and vermiculite have a high sorption for cesium, whereas montmorillonite and especially caolinite have lower sorption capacities (Coleman *et al.* 1963, D'Souza *et al.* 1980). According to Schultz (1965), two cesium fixation mechanisms can be present in soil, depending upon whether this elements is present in large or in very small quantities. Cesium fixation in clay minerals takes place at only a small number of sites, situated in the interlayer-edge zones. When intercepted in these sites, radiocesium is gradually immobilized into interlattice positions leading to a decrease in availability. Tamura & Jacobs (1960) suggested that the binding of cesium by clays can result either from "edge fixation" or

from "interlayer fixation". Failure of attempts to correlate cesium uptake by plants with clay content of the soil (see Sandalls & Bennett 1992) could be due to a non-differentiation of the various types of clays, or to non-differentiation of the various adsorption sites in those clays. Brouwer *et al.* (1983) showed three kinds of adsorption sites to be associated with the ubiquitous hydrous mica, illite; these sites correspond to about 0.5%, 3% and 96.5% of the total exchange capacity, and each site showed a characteristic selectivity coefficient (see also Cremers *et al.* 1988). There is now a general consensus that the selective sorption of radiocesium in soils is related to the action of micaceous clays (Tamura & Jacobs 1960, Sawhney 1972, Aston & Duursma 1973, Francis & Brinkley 1976, Brouwer *et al.* 1983). A key factor in the fixation process is played by the low hydration energy of ions such as Cs^+ , Rb^+ , K^+ and NH_4^+ . These specific effects are thought to take place at the edges of clay particles, characterized by partially expanded layers, generated by weathering and possibly by the action of large hydrated cations such as Ca^{++} and Mg^{++} (Comans *et al.* 1991). These sorption sites are often called Frayed Edge Sites (Wauters *et al.* 1994).

Many authors reported that most of the radiocesium deriving from the Chernobyl accident is still localized in the upper organic horizons years after deposition. However, "old" cesium originating from nuclear weapon tests was often encountered in deeper soil layers. Colgan *et al.* (1990) found that in Irish pastures over 90% of Chernobyl radiocesium remained in the 0-10 cm section, while 17% of weapon fallout radiocesium lies below the 10 cm horizon. This indicates that, although slowly, radiocesium eventually can reach the mineral horizons. This fact is supported by Rühm and collaborators (in litt.), who suggest to look at the time behaviour of the Cs-137/Cs-134 ratio in different soil horizons. In the L- and Of-horizon of a coniferous forest in South Germany, these ratios are practically constant with time, showing values close to the initial value of 1.75 characteristic for the Chernobyl fallout near Munich (Hötzl *et al.* 1987). From 1987 to 1994, the ratio is decreasing from 1.95 to 1.85 in the Oh-horizon, from 2.5 to 2.0 in the Ah-horizon, and from 2.7 to 2.2 in the B-horizon. This is due to the fact that Chernobyl cesium with a ratio of 1.75 is entering these deeper soil horizons. There, mainly weapon fallout cesium is present, containing almost no Cs-134, and, accordingly, a comparably higher ratio is expected.

The relative abundance of clay and mica

minerals, particularly illite, in deeper soil horizons results in the rapid and near irreversible immobilization of cesium in the soil (Davis 1963, Andersen 1967, Cataldo 1979, Cremers *et al.* 1988). Soil/plant transfer of radiocesium is inversely correlated with the clay content, and positively correlated with cation exchange capacity and soil organic matter (Frederiksson *et al.* 1969, Van Voris *et al.* 1990). Thus, forests with clay-poor soils might have higher radioactivity in the over- and understory than forests with clay-rich soils, the transfer rate being, however, equally dominated by the amount of radiocesium contained in the organic horizons.

In conclusion, radiocesium activities show distinct vertical patterns in soils of natural ecosystems. As different plants exploit different soil horizons, part of the high variability in radioactivity data in wild plants can be attributed to vertical soil gradients.

Biological factors

Absorption and re-deposition by plants - Root or mycelial uptake of radiocesium in the soil may be a cause of cesium depletion in certain soil horizons. However, this phenomenon has been never quantified. Some aspects related to re-deposition by plants through leaching and litter fall were already considered in the previous chapters. A quantitative estimate of the relative importance of litter fall and leaching in the transfer of radiocesium from the plant to the upper soil layers is not easy, because there are relatively few data on the contribution of leaf fall to soil radioactivity. Bunzl *et al.* (1989b) calculated that, two years after direct deposition in a spruce stand, falling needles accounted for c. 7% of the quantity of radiocesium transferred from the canopy to the ground. Direct comparisons among representative stands of the major holarctic biomes are not available for radiocesium, but there are many data on other elements, especially potassium, which were summarized by Rodin & Bazilevich (1967). In Boreal forests of Germany potassium leaching accounts for 8-10 kg/ha a year (Ehwald *et al.* 1961), whereas Magdwick & Ovington (1959) give the somewhat higher value of 21.4 kg/ha for coniferous stands in Britain. In temperate deciduous forests Magdwick & Ovington (1959) calculated a leaching range for potassium of 22-27 kg/ha. As far as litter fall is concerned, the principal organogen elements (Ca, K, P, S) account for 60-80% of the total ash elements deriving from litter fall in coniferous forests, their absolute content varying from 20 to 30

kg/ha in pine forests, and from 9 to 90 kg/ha in spruce forests. The return of ash elements with litter fall is much greater in deciduous forests, the main organogen elements ranging from 150 to 260 kg/ha, with 70-90% coming from leaf fall (Rodin & Bazilevich 1967).

Biological fixation in soils - A rather surprising result of many post-Chernobyl radioecological studies on natural ecosystems is the long-lasting persistence of radiocesium in organic horizons. Many pre-Chernobyl studies, carried out in agroecosystems with poorly developed organic horizons, attributed to clay, and not to organic matter, an important role in retaining radiocesium and preventing its downward migration through the soil profile. Organic horizons have a low sorption capacity; hence, it should be expected that radiocesium migrates into the mineral horizons rather rapidly, especially in the presence of an additional mean proton by acid rain. The reason for the very slow migration of radiocesium from organic to mineral layers is still not well understood. Since the actual migration pattern is contrary to that expected, other mechanisms than soil sorption processes may be of greater influence on radiocesium migration. A factor which certainly has been underestimated by many radioecologists, is that in organic horizons radiocesium can be immobilized by the soil microflora and -fauna. The organic and the intermixed organic-mineral horizons are densely rooted by the fine roots of trees and understory plants, most of which are in symbiosis with mycorrhizal fungi, and host an intense microbial activity. According to Clint *et al.* (1990) litter decomposition and radiocesium release are likely to be influenced by the nature and level of microbial activity in the soil. Microbial decomposition accounts for 80-90% of the total litter decomposition (Armson 1977). The rate of decrease of radionuclides from the forest litter depends on several factors, including temperature, moisture and type of foliage. Clint *et al.* (1990) found that the radiocesium recycling rate increases in environments where frequent wet and dry cycles occur, compared to stable environments where the litter is perennially moist. Thus, a major temporary reservoir for radiocesium in forests is organic litter. The corresponding transfer rate is equivalent to the biological half-time for organic matter. According to Armson (1977) litter from deciduous trees begins to release water-soluble ions on the first day, while conifer needles release ions at a much slower rate.

The decomposition time of litter is of c. 6 months in Tropical forests (Pachman & Harding 1982), c. 3-5 years in Temperate forests (Schell & Myttenaere 1989) and more than a decade in Boreal forests (Pachman & Harding 1982).

Some authors have introduced the concept of the "rhizoplane population", which is defined as the bacterial, and possibly the actinomycete population living on the outer surface of the root. There is a continuous gradation in the characteristics of the microbial population living on the root surface, in the soil, but close to the root, and in the soil well away from the root (Russell 1965). Microbial cells play a significant role in the accumulation, cycling and transfer of heavy metals in the soil (Gadd 1988; Hughes & Poole 1989), and a considerable part of the potential plant nutrients in soil may be more or less firmly bound in the fungal biomass (Bäath & Söderström 1979). The quantity of fungal hyphae found in a forest soil, depending on the species, ranges from 3 to 10 km/g soil, and about 30% of the total soil content of radiocesium has been estimated to be incorporated in fungal hyphae (Olsen *et al.* 1990). Guillitte *et al.* (1990c) carried out experiments on undisturbed soil samples in controlled chambers. Half of the samples were irradiated to kill soil organisms. After two weeks of artificial rain, the radiocesium activities in the irradiated samples had dropped by 38% compared to non-irradiated samples. This percentage corresponds to the theoretical percentage of cesium which can be fixed by soil fungal biomass (Olsen *et al.* 1990). Very similar results were obtained by Guillitte *et al.* (1994) in an experimental study carried out in a Swedish Boreal forest; these authors found that c. 5% of the deposited radiocesium was retained in roots, and c. 40% was fixed in fungal mycelia.

Considerable amounts of radiocesium could be also fixed in the bacterial biomass, with phenomena of short-term release and recycling within the bacterial populations of the soil. Although the nutrient incorporation of fungi and bacteria is highly variable (Anderson & Domsch 1980), it is very probable that the release of nutrients and radiocesium may be retarded by retention of the elements by soil micro-organisms in a process referred to as microbial immobilization (Witkamp & Barzanskyi 1968). The effects of temperature, rainfall and fauna on the transfer of radiocesium related to the consumer-decomposer microcosms was studied by Witkamp & Frank (1970). The high activities in plants and mushrooms which take up the nutrients preferably from the organic layers suggests

that, being incorporated in organisms, a physical migration could be effectively prevented. This is contrary to agricultural ecosystems, where cesium is mainly fixed by clay minerals. There, cesium is hardly available for plants, which corresponds with the low activities of agricultural products and the low migration rates reported in the literature.

The role of microorganisms in the fixation of elements in organic horizons of forest soils still needs further studies, as it is indicated by the contradictory results obtained by different authors. For example, Ouijja & Myttenaere (1994), in an experimental study carried out in the laboratory on Belgian forest soils, concluded that the microflora contributes only weakly to the immobilization and retention of radiocesium in forest soils. On the other hand, Bruckmann & Wolters (1994), studying several different forest-types in Germany, found that the microflora strongly contributes to the immobilization of radiocesium in forest soils. By transporting nutrients into the upper horizons (above the roots of trees) the microflora significantly accelerates the recycling of radiocesium in forest ecosystems. According to Bruckmann & Wolters (1994), both immobilization and transport of radiocesium are determined by three major factors: radiocesium availability, growth conditions of the microflora and biotic interactions.

Besides micro-organisms, part of the cesium found in the organic horizon could be biologically fixed in fine roots, which are normally not separated from the soil in radiological measurements. Coughtrey *et al.* (1990) demonstrated that in British pastures radiocesium specific activity in fine roots was higher than in soil. Padovani *et al.* (1990), working in northeastern Italy, found that retention of radiocesium is highest in the top soil layer and decreases with depth; they related this buffering effect to the greater soil biological activity and root density. A similar observation was made by Bachhuber *et al.* (1982) with regard to the distribution coefficient of radiocesium in German soils. According to Merish (1990) c. 10-20% of soil radiocesium could be retained in root systems of trees and herbaceous plants.

There are relatively few studies dealing with the effect of the soil fauna on the vertical migration of radiocesium. Invertebrates have been shown to significantly affect the microbial immobilization of nutrients in beech forests on acid soils (Scholle *et al.* 1992, 1993, Bunzl & Schimmack 1988). Caput *et al.* (1990), studying permanent pastures, found that the cesium which was initially concentrated in the 5 cm

top layer moved down gradually at a rate larger than expected, independently of soil properties and probably in relation to the mixing action of earthworms. Bruckmann & Wolters (1994) found that the microbial transport of radiocesium into the L layer was accelerated by the mesofauna (microarthropods and enchytraeids), which suggests that immobilization and recycling of radiocesium by the microflora in the organic layers of forest soils is significantly affected by edaphic invertebrates (see also Wolters 1991 a,b). Llauro *et al.* (1994), working in evergreen Mediterranean forest ecosystems of Spain, suggested that faunal mixing activity is an important mechanism for radiocesium distribution in Mediterranean soils (see also Sauras *et al.* 1994).

UPTAKE OF RADIOCESIUM BY PLANTS

In a complex forest ecosystem, different plant groups may uptake radiocesium from different sources. Atmospheric deposition is the main source of uptake for bryophytes and lichens, an important source for higher plants, an occasional source for most fungi. Contrary to bryophytes and lichens, higher plants take up radiocesium also from the soil, through the root systems (and eventually through mycorrhizal symbiosis). Saprophytic soil fungi mainly exploit the radiocesium pool in the soil and/or in dead organisms. A special case is that of parasites (mainly fungi, but also some higher plants), which take up radiocesium from living organisms. The ecological interpretation of radioactivity values in lichens, mosses and most soil fungi might be conceptually easier than that relative to higher plants, since these organisms take up radiocesium from a single main source. For this reason, they may be most profitably used as bioindicators of environmental radioactivity. The case of higher plants is much more complex, as it is not always easy to differentiate the contribution of foliar uptake, root uptake, and uptake from mycorrhizal fungi to the total radiocesium pool inside the plant. In the following chapters, the main uptake mechanisms for different systematic groups are briefly reviewed.

CRYPTOGAMS

Mosses

Bryophytes are of little interest for radiation protection studies, as they do not constitute an important element in the forest food chain. However,

they are an interesting object for radioecological research. In several forest ecosystems, and especially in forests of rainy areas, bryophytes are an important element of the total biomass. The thick carpets of bryophytes covering large parts of the forest floor can intercept a great quantity of the total deposition, slowing down the transfer to soil.

Mosses are able to trap and retain a large share of wet radioactive deposition (Sawidis 1987), since: a) they lack epidermis and cuticle, b) they have a large surface-volume ratio, and a high absorbing power, c) they do not possess roots, and accumulate mainly in a passive way, e) they lack a well-developed vascular system, so that there is but little transport of radionuclides within the plants. Mosses acquire most nutrients and elements from liquid deposition, by capillarity in the spaces between leaflets, and between leaflets and stems. According to Rühling & Tyler (1971), they are effective collectors of most metals, acting also as ion exchangers and metal-organic complexing agents. Mosses have large ion exchange and chelating capacities and retard the transport of radionuclides under natural conditions. Interspecific differences were noted in the interception and retention of airborne particulates; these seem to be mainly due to physical characteristics such as surface morphology and the degree of local shelter, and it has been speculated that passive particulate trapping is the dominant uptake mechanism in most bryophytes (Boileau *et al.* 1982). The absorbing power of several species was studied by Mägdefrau & Wutz (1951); widespread and common species such as *Ptilidium crista-castrensis*, *Pleurozium schreberi* and *Dicranum undulatum* absorb completely more than 1 cm of liquid precipitation in a very short time; considering also the lower part of the bryophyte mats, constituted of dead leaflets and stems, the absorbing power rises to 3 cm of precipitation. Other species, such as *Mnium* spp., and *Polytrichum* spp., absorb less, due to a different morphology, and especially to the scarcity of capillary spaces between leaflets; the water is slowly released by evaporation within several days. The intensity of precipitation has an influence on interception rates and removal half-times.

High concentrations of radiocesium in bryophytes were reported by Horrill *et al.* (1990). The residence time of radionuclides has been estimated to be quite long prior to the input of acid precipitation (Schell & Tobin 1990); however, an excess of acidity in many regions has altered the ion exchange capacity and speciation of elements

causing greater mobility in the forest floor. Estimates of the removal half-life in some moss species range from c. eight months in Great Britain (Sumerling 1984) to 18 months in Belgium, with a linear decrease of radioactivity during the first years after deposition (Guillitte *et al.* 1990b). On a regional scale, Giovani *et al.* (1994) used the moss *Ctenidium molluscum* as a bioaccumulator for drawing a radiodeposition map of NE Italy; the carpets were shown to be able of intercepting most of the radiocesium deposited after the Chernobyl accident, and the calculated removal half-time was of c. 46 months. Below the local scale, Guillitte *et al.* (1989) demonstrated that bryophytes can be efficiently used to map deposition micropatterns and water circulation from the canopy to the ground within forest communities.

Lichens

Lichens are efficient collectors of atmospheric trace-element pollution, owing to their relatively high accumulation capacity, and to their dependency on nutrients from the atmosphere (Tuominen & Jaakkola 1973, Rao *et al.* 1977, Eckl *et al.* 1984). Some studies even concern lichens and radioactivity deriving from satellite crashes (Taylor *et al.* 1979), geothermic radioactivity (Matthews 1981), or radioactivity deriving from Uranium mining (Pettersson *et al.* 1988). However, most of the research on radionuclides in lichens was carried out in the tundra biome, well before the Chernobyl accident, chiefly because of the fallout from atmospheric nuclear weapons testing on the tundra biome, and of its effects on the food chains lichen-caribou-man (Hanson 1982, Pruitt 1963, Aberg & Hungate 1967, Persson 1967), and lichen-caribou-wolf (Holleman & Luick 1978, Holleman *et al.* 1980). With a few exceptions (e.g. Kwapulinski *et al.* 1985a,b), it is only after the Chernobyl accident that lichens of the Temperate zone were studied in some detail.

The first radiocesium determinations in lichens were made by Gorham (1959), followed by Hviden & Lillegraven (1961), who reported that lichens are much more efficient in accumulating radiocesium than higher plants. The first large-scale investigations were those of Lidén (1961) and Weichold (1961). Analyses of radiocesium content in lichens from Finland, the U.S.S.R. and Alaska, as determined by Paakola & Miettinen (1963), Salo & Miettinen (1964), Rakola & Miettinen (1971), Niznikow *et al.* (1969), Hanson (1967, 1971) and

Martin & Koranda (1971) showed a wide range of variation, with values ranging from 1000 to 2500 Bq/Kg-1 dry weight. Svennson & Lidén (1965) and Hanson (1967) found a good correlation between total radiocesium content in lichens and total estimated deposition, which led to the use of these organisms as biomonitors of radioactive deposition. Similar results were also obtained after the Chernobyl accident: Sloof & Wolterbeek (1992) found the same relation between radiocesium activity in lichens, expressed on a dry weight basis, and that in soils, expressed per surface unit, in several samples of *Parmelia sulcata* collected in the Netherlands. Very striking is also the similarity between the map of radiocontamination of Swedish soils and the analogous map for lichens (Jones *et al.* 1989).

In the early 60's the radiocesium content in Finnish lichens was 5 to 10 times higher than in vascular plants (Salo & Miettinen 1964). After 1965 there was a rapid decrease of nuclear weapon tests fall out, but this was not followed by a corresponding decrease in lichens (Plummer 1969, Tuominen & Jaakkola 1973). Lichens have been frequently used to monitor spatial patterns in radioactive deposition over wide areas (see e.g. Feige *et al.* 1990). Not only Arctic terricolous lichens can be effectively used as biomonitors; an example is given by Sloof & Wolterbeek (1992), which used also epiphytic species as suitable indicators in Holland.

According to Kwapulinski *et al.* (1985a,b), radiocesium uptake is generally highest in terricolous, lowest in epiphytic lichens, epilithic species ranging inbetween. This might depend on several factors, such as the inclination of the thallus and its hydration physiology. Guillitte *et al.* (1994), for example, found that lichens with a horizontal thallus occurring on tree branches were twice as contaminated as those with a vertical thallus growing on tree trunks. Most of the radiocesium is deposited at the thallus surface, whereas uptake from the soil seems to be negligible; according to Niznikov *et al.* (1969), only 2% of the soil radiocesium can penetrate into terricolous lichen thalli. Hanson & Eberhardt (1971) found a seasonal cycle of radiocesium in lichens, with maximum values in summer and a minimum in midwinter.

The distribution of radiocesium in lichen thalli was the object of several studies, starting from the early 1960's. In fruticose lichens the apical parts of the thalli contain 2 to 14 times more cesium than the basal parts (Paakola & Miettinen 1963, Hanson

1967). The mobility of radiocesium inside the thallus was studied by Nevstrueva *et al.* (1967); the results indicate that Cs and Sr are rather mobile within the thallus, Cs being less leachable. Carpets of *Cladonia stellaris* in Sweden studied from 1986 to 1990 showed that there was a slight downward movement of radiocesium through the lichen carpets; however, some 70-80% of radiocesium still resided in the upper 3 cm (see also Kreuzer & Schauer 1972, Mattsson 1974). According to Hanson & Eberhardt (1971) the concentrations of radiocesium are relatively stable in the upper parts of terricolous lichens, but the radionuclide is apparently cycled between the lower portions of the lichen mats and the humus layer. Feige *et al.* (1990) present autoradiographies of *Cetraria islandica* and *Cladonia arbuscula*: the radionuclides are almost uniformly distributed throughout the thalli, although the upper parts of *Cladonia arbuscula* appear to be more radioactive than the lower parts. In *Cetraria islandica*, the apothecia tend to accumulate more radionuclides than the rest of the thallus. Some of the pictures show also the presence of "hot spots" corresponding to products of nuclear fusion or to highly radioactive particles deriving from the Chernobyl accident, and trapped inside the thalli. The same authors have also tried to wash the lichens in deionized water: after a week only 8% of the radionuclides were removed, and after two weeks the removal interested only 3% of the remaining radioactivity.

Morphological differences between species may play an important role in their capacity to intercept and retain radiocesium. Kwapulinskyi *et al.* (1985a,b) found species-specific differences in four species of *Umbilicaria* collected in Poland. An interesting methodological problem has been addressed by Sloof & Wolterbeek (1992); these authors studied radiocesium accumulation in a foliose lichen, *Xanthoria parietina*, and expressed the activity on a weight and on an area basis. They found that in the former case large variations were evident between parts of the thallus with and without fruitbodies, whereas the average radiocesium activity expressed per surface area was almost constant. In general, lichens, especially foliose and fruticose species, have a high surface area to mass ratio; this property is often reported as one of the main reasons for their relatively high capacity to accumulate heavy metals and radionuclides (see e.g. Seaward *et al.* 1988, Nimis *et al.* 1993). According to Seaward *et al.* (1988), however, the high variability in radiocontamination observed in

different species growing together are due more to ecological factors than to physiological or morphological differences among species. Like in higher plants, uptake and release of cesium in lichens may be affected by the chemically related and physiologically important elements potassium, sodium, and, in a lesser degree, calcium (Tuominen & Jaakkola 1973). This factor, however, seems to be important only on a physiological level.

Much less studied are the physiological mechanisms underlying radiocesium uptake by lichens. According to Tuominen & Jaakkola (1973) some process of cationic exchange should be involved. However, Handley & Overstreet (1968) demonstrated that the fixation of radiocesium in lichen thalli does not depend on their physiological activity, being mostly a passive phenomenon. According to Subbotina & Timofeev (1961), however, radiocesium ions were still strongly bound and difficult to remove from partially decomposed thalli. This would suggest that the ions are transported into the thallus and bound to cytoplasmic molecules through processes of active translocation. There is some evidence that lichens are more resistant than other organisms to high radioactivity: according to Biazrov (1994), lichen thalli measured near Chernobyl showed extremely high radioactivity values, but these did not cause any visually discernible anomalies in the development of lichen thalli, confirming the data on the high resistance of lichens to radioactive irradiation earlier presented by Brodo (1964).

The biological half-time of radiocesium in lichens is very variable, depending on the species, and especially on precipitation (see Tuominen & Jaakkola 1973). The literature values range from 2.7 to 17 years. The effective half-life of radiocesium in carpets of *Cladonia* was estimated differently by different authors: from 5-8 years, to 17 ± 4 years, and 7-8 years in the upper 3 cm, and about 8-10 years in the whole carpet, (Ellis & Smith 1987, Lidén & Gustafsson 1967). Martin & Koranda (1971) gave a biological half time of c. 8 years in interior Alaska, of 3-3.7 years in coastal areas. These differences might be due to differences in precipitation between the humid coastal areas and the relatively dry internal regions. Lidén & Gustafsson (1967) and Niznikov *et al.* (1969) suggested that as time elapses from the moment of deposition, the effective half-life of radiocesium for lichens will increase. In Canada, after the cessation of nuclear weapons' testing in 1962, the cesium deposited as fallout was available to agricultural plants for only a few years

(Bird 1966, 1968); further north, the fallout was not lost as quickly; lichens, mosses and vascular cushion plants between 60° and 70° N demonstrated significant available Cs-137 in the 1980's, long after it had disappeared from the more contaminated regions further south (Hutchinson-Benson *et al.* 1985, Meyerhof & Marshall 1990). According to Hanson (1967), the biological half-life period in *Cladonia stellaris* is of 3-6 years when deposition has happened in the liquid form, of 1-13 years when it has occurred in the gaseous form. Different formulas to calculate the removal half-times in lichens were proposed (e.g. Gaare 1990, Sloof & Wolterbeek 1992). However, a generalization is probably difficult: different factors affect the actual half-life of radiocesium in lichens; some of them depend on features of the lichen itself, such as growth rates, genetic variability, density of fructifications, others depend on characteristics of the station, such as microclimatic variability, leaching of the substrata, geographic situation, etc. The sampling techniques, as well, may have an influence on the estimates: different values might be obtained if sampling the upper vs. the lower parts of the thalli.

Fungi

Pre-Chernobyl studies - The first studies demonstrating a high capacity of mushrooms of absorbing relevant quantities of radionuclides date back to the early 1960's (Marah *et al.* 1962, Kiefer & Maushart 1965). Grüter (1964, 1966, 1971) found much higher activities in fungal fruitbodies than in other forest plants. A study by Maushart (1966) showed that radiocesium activity in the human body within selected populations doubled during Autumn, and that this was mainly due to a high intake of mushrooms. Pre-Chernobyl radioecological data concerning the radiocontamination of mushrooms were rather contradictory: some authors found significant differences among species (Grüter 1966, Maushart 1966, Haselwandtner 1978), others claimed that there were high differences also among individuals of the same species growing on the same soil (Röhleder 1967, Seeger & Schweinschaut 1981). There was a certain agreement on the relations between radiocesium activities in mushrooms and type of soil, the values being highest in sandy (Grüter 1964, Maushart 1966, Röhleder 1967) or organic soils with low pH (Johnson & Nayfield 1970). Ijpelaar (1980), however, claimed that the large differences in radioactivity were mainly due to

species-specific differences, and not to the type of soil. One of the largest pre-Chernobyl surveys of radioactivity in mushrooms was published by Seeger & Schweinshaut (1981), who studied more than thousand samples of European fungi. They also found a very high degree of variability. Marked fluctuation of the cesium content was observed even in fungi of the same species grown at the same location.

Before Chernobyl, there were only a few efforts to relate radioactivity data in fungi and ecological factors. Röhleder (1967) claimed that mycorrhiza and lignicolous fungi generally show lower radioactivity than saprophytic fungi, but these data were not confirmed by those presented by Seeger & Schweinshaut (1981).

Post-Chernobyl studies - Immediately after the Chernobyl accident high radioactivity levels were measured in mushrooms from several European countries (e.g. Nimis *et al.* 1986, Kuyper 1987, Mascanzoni 1987, Rückert & Diehl 1987, Byrne 1988). High contamination levels in deer meat, recorded in Canada, were attributed to a diet chiefly consisting of mushrooms by Meyerhof & Marshall (1990); seasonal variations in the radiocontamination of wild mammals has been often attributed to a diet rich in fungi in certain periods of the year (e.g. Karlén *et al.* 1991, Lindner *et al.* 1994). Thus, fungi have received considerable attention after the Chernobyl accident, as an important source of radioactivity to animals and to man. The first large-scale survey of mushroom radiocontamination after Chernobyl was carried out by Nimis *et al.* (1986) in NE Italy. Significant differences in radiocesium concentrations were found both among species collected in the same stations, and among average values of the stations themselves. The differences among species of the same stations were attributed to the depth of the mycelium in the soil; the concentrations were highest in saprophytic fungi living on the litter layer, and hence with superficial mycelium, lowest in the symbionts with deep-rooting deciduous trees. The differences among stations were due to differential deposition within the survey area. This allowed the elaboration of a contamination map of NE-Italy based on data relative to saprophytic fungi, that was in very good agreement with later data on the deposition values in the area. Very similar results were obtained by Guillitte *et al.* (1987) in an analogous study carried out in Belgium. Already in 1987, the radioactivity of mycorrhizal fungi tended to increase, that of

saprophytic fungi to decrease (Nimis *et al.* 1988b, Giovani *et al.* 1990, Römmelt *et al.* 1990), as a consequence of the slow migration of Chernobyl radiocesium from the upper to the lower parts of the organic layers (Nimis *et al.* 1988b, Molzahn *et al.* 1990). Guillitte *et al.* (1994) discussed the differences between obligately and facultatively mycorrhizal species: since facultative mycorrhizal species depend more on dead organic matter, they tend to develop their mycelia in holorganic horizons, whereas obligate species are able to explore deeper soil layers; the fact that these deeper layers are less contaminated and have a lower radionuclide bioavailability could account for the lower contamination found in obligate species (see also Heinrich 1992). The extremely unequal distribution of radiocesium in the soil profiles just following deposition represents a rather special case, and species-specific differences in radiocesium absorption cannot be excluded for explaining differences in radiocontamination among species. Kammerer *et al.* (1994) and Rühm (pers.comm.) measured Cs-134 and Cs-137 activities in mushrooms and in the corresponding soil horizons. They found that the ratio of Cs-134 to Cs-137 varies within one soil horizon and in mushroom species on the same site with certain ranges, and that the mass of saprophytic and mycorrhizal mycelia should be localized in organic horizons. The large variations in radiocesium activities, especially found in symbiotic mushrooms, were explained less by mycelium depth than by physiological differences. Each species seemingly has a certain accumulation rate, which is indicated by the ratio of the absolute Cs-137 activities in different mushrooms being about the same at different sites. Yoshida & Muramatsu (1994), studying radiocontamination of mushrooms in Japan, found that, apart from species-specific differences in accumulation for some ammonia fungi (genus *Hebeloma*), the concentrations of radiocesium in mushrooms reflected those in the soil layers in which the mycelia were growing, with generally higher values in mycorrhizal than in saprophytic fungi.

However, according to Guillitte *et al.* (1994) no explanation has been provided to date for the large contamination differences in taxonomically and ecophysiologically similar species, such as mycorrhizal fungi of the genus *Boletus* s.lat. These authors suggested that, as far as mycorrhizal species are concerned, radionuclides are blocked differentially in the mycelial sheath surrounding the roots, depending on the fungal species.

Physiological aspects - Seeger & Schweinschaut (1981) found that in single fruit-bodies, the cesium content was usually highest in the flesh of the cap, lowest in the gills, or, more rarely, in the stem; radioactivity in the stem is usually no more than 48% of the cap on average (Fraiture *et al.* 1989, 1990). Comparable results were obtained by Rückert & Diehl (1987), Heinrich (1987), and Heinrich *et al.* (1989). Rückert & Diehl (1987) and Bakken & Olsen (1990) measured the tubes of *Boletus* separately from the stems, and found that they displayed an activity 50% to 100% higher than those of caps minus the tubes. This could be explained by the fact that radiocesium within the fungal carpophores behaves similarly to potassium (Steinberg 1946), the potassium content being generally highest in the caps (see Seeger 1978). Aumann *et al.* (1989) found that badione and norbadione, two pigments present on the cuticle of the cap of *Xerocomus badius* and related species, are able to complex radiocesium, which can result in a considerable accumulation of this isotope (see also Mietelski *et al.* 1994). Molzahn *et al.* (1990) measured a much higher activity in the cuticle than in the flesh of *Xerocomus*.

Little is known about the requirements of mushrooms for mineral elements; high potassium contents were reported in several species (Seeger 1978), and, according to Mascanzoni (1990), a competition Cs-K similar to that found in many plants might be expected, with high radiocesium uptake reflecting potassium-poor substrata. Guillitte *et al.* (1994), studying the mushrooms of a Swedish Boreal forest, found that the highest radiocesium concentrations were found in members of the *Gasteromycetes*, a taxonomic group characterized by extremely low carpophore potassium content, whereas the highest concentrations were reached in members of the *Cortinariaceae*, which exhibit the highest potassium concentrations. Other authors, however, observed that the cesium content in mushrooms is very variable, while that of potassium is almost constant (Eckl *et al.* 1986, Byrne 1988, Mascanzoni 1990). This is not necessarily related to differential absorption of the two ions, as the main cause is likely to be the unequal repartition of cesium in the soil, and the different mycelium depths of different fungal species. According to Andolina & Guillitte (1990) an important cesium fraction is bound to lignin in the holorganic horizons; the higher radiocontamination values of mushrooms could be partly explained by their ability to decompose lignin; some of the cesium unavailable to

other plants would be available to mushrooms.

The mechanisms involved in the cesium accumulation in fungi are not known, but a plausible hypothesis is that it depends on properties of the transport enzymes in the cell membrane as to the discrimination between Cs^+ and K^+ ions (Byrne 1988). According to Oujia & Myttenaere (1994), however, cesium is probably accumulated in the fungal biomass as Cs^+ , and not complexed with organic compounds. In contrast to the well-known preference of mushrooms for K versus Cs (Conway & Duggan 1958), experiments carried out by Olsen *et al.* (1990) demonstrated that pure cultures of mycorrhizal fungi did not show any preference for K versus Cs, and that in some species there was even a preference for Cs versus K. The accumulation of radiocesium is significantly affected by species-specific features and the variation in uptake values can be due to the affinity of Cs-transporting carriers in the plasmamembrane of the hyphae to this alkali metal (Rothstein 1965), or to differences in cell wall structure (De Rome & Gadd 1987).

Transfer Factors - One of the problems in calculating transfer factors for fungal carpophores lies the fact that fungal mycelium spreads widely, both horizontally and vertically in the soils. Mascanzoni (1990) suggested as a transfer factor for fungi the ratio between the activity in the fruit body (expressed as Bq/kg fresh weight) and the activity deposited on the soil surface, expressed in Bq/m². Such a transfer factor can be criticized from several points of view. First, in mushrooms the expression of the activity in terms of fresh weight may be a relevant source of noise; Mascanzoni (1990) claims that the expression in terms of dry matter can be easily estimated considering that average water content of mushrooms is about 90% with small variations; this, however, is true only for mushrooms in an optimal hydration state; during even brief periods of relative drought the water content of mushrooms may be considerably less, which has an obvious influence on the measurement. Guillitte *et al.* (1990c) and Lambinon *et al.* (1988) developed an approach for estimating the depth at which mycelium development occurs, and, therefore, for identifying transfer factors from the soil layers that are actually colonized by fungi. The method is based on the estimation of the Cs-134/Cs-137 ratio in fruitbodies and in different soil horizons, and on the assumption that the ratio observed in a fungus should be identical to that in the layer of soil exploited by its mycelium. Transfer factors could be calculated

on the basis of the radioactivity of the soil horizon in which most of the mycelium occurs. This method, however, can be applied only a short time after a relevant direct deposition, due to the short half-life of Cs-134; problems in its application may arise already a few years after deposition, also due to the progressive disappearance of sharp vertical patterns of radiocesium within organic horizons (Kammerer *et al.* 1994).

Due to the complex pattern of fungal hyphae in natural ecosystems, transfer factors for fungi are best calculated in the laboratory, on pure fungal cultures. Ouijja & Myttenaere (1994) found clear differences in the transfer factors of radiocesium among different fungal species.

Fungi as bioindicators - The use of mushrooms as bioindicators or bioaccumulators of radioactive pollution is often hampered by the high variability of field data. Furthermore, most macrofungi can be collected only during short periods, and the production of carpophores is not regular in time. The study of radiocontamination of mushrooms gives a good example of the progress of radioecology in post-Chernobyl times. The initial high variability in the data has been mostly explained, up to an extent that it is now possible to utilize mushrooms as bioindicators of radioactive deposition over vast areas (Nimis *et al.* 1986, Haselwandtner *et al.* 1988, Nimis & Cebulez 1989, Nimis 1990). Andolina & Guillitte (1990) found a good correlation between available cesium in the organic horizons and the contamination of *Xerocomus chrysenteron*. A radiocontamination map of NE Italy was published by Nimis *et al.* (1990) using saprophytic fungi as bioindicators. Radiocontamination maps of Poland were published by Mietelski *et al.* (1994) using *Xerocomus badius* as a bioaccumulator. A list of methodological precautions to be taken when using fungi as bioindicators has been published by Fraiture (1992).

VASCULAR PLANTS

Several authors have reported that different species of vascular plants, even from the same ecosystem, take up radiocesium at different rates. Horrill *et al.* (1990) found large variations in the concentration ratios of radiocesium of higher plants in Cumbria, and a poor correspondence between plant and soil activities. Nelín & Nylén (1994) found that the correlation between the fall-out cesium on the ground (Bq/m²) and the concentration in/on

plants (Bq/kg) was more significant during the first year after the fall-out, probably due to direct contamination. In the following years, biological and other ecological factors influencing root uptake and translocation inside the plant become more important, and tend to mask the correlations between radiocesium concentrations in plants and in soils. According to Cooper & Mattie (1990), the determination of concentration factors for radionuclides taken up by trees in natural environments is difficult, due not only to the heterogeneous distribution of radionuclides in the soil, but also to the variable uptake of individual trees, as the concentrations of radionuclides vary with the tissues examined, and within individual tissues. Wirth *et al.* (1994), studying two coniferous forests in Bavaria, found very poor correlations between radiocesium concentrations in plants and in the underlying soil. High interspecies variability in uptake rates and poor correlations with soil parameters suggest the existence of species-specific differences in cesium uptake which would render any modelling effort at ecosystem level almost impossible, as it would be quite difficult to predict the levels of radiocesium in vegetation from that in the soil. There are, however, other authors, such as Sandalls *et al.* (1990), and Nimis *et al.* (1990a,b, 1994), who found significant correlations between radiocesium activities in different species and some relevant soil parameters, which suggests that species-specific, physiologically based uptake differences may have little influence on the levels of radiocesium in vegetation, and that, instead, a major role is played by ecological factors.

In the following we shall mainly focus on some main mechanisms of root uptake, and on other factors related to the physiology of the plants, which might influence radionuclide concentrations in plants. We anticipate that in our opinion most of the variability found is due to lack of consideration of relevant parameters, that the causes of variability can be largely understood, and that predictive models based on these relevant parameters are possible.

Uptake via leaves

Foliar deposition is potentially a major source of food chain contamination by radioactive substances (Russel 1965, Muller *et al.* 1983). After direct deposition on foliar surfaces, the contaminants can be transferred directly to animals feeding on the foliage. There is ample evidence that the uptake of

nutrients occurs also through foliar absorption. According to Muller *et al.* (1983) elements absorbed by leaves are utilized by the plant and the excess may be relocated in the phloem, to other growing leaves, or even to roots. The amount of foliar contamination depends on the growing stage of the vegetation, and is greatly influenced by leaf morphology (which also affects interception ability) and by climatic conditions (rainfall, wind speed and direction, etc.). Broad, hairy leaves, or those with evident microstructures on the leaf surfaces have a higher interception power. Rauret *et al.* (1994) found a considerably higher retention of aerosols on young than on old leaves of *Quercus ilex*, due to the higher density of trichomes on the adaxial surface of the former, and also to the position of the leaves on the tree; these authors also concluded that neither the abiotic layer nor the cuticle of the leaves play an important role in the retention of cesium; more than 95% of cesium is soluble; 45% of the deposited cesium remains on the leaf surfaces but is solubilized by water, while 55% is retained by the leaves.

When the deposited particles are wetted by precipitation, radiocesium ions are adsorbed by the epidermal cells. Bukovac *et al.* (1965) differentiate between a passive diffusion in the "apparent free space" of the cell wall, and an active transport through the plasmalemma. Once the symplastic barrier is overcome, there may be further transport, up into cell vacuoles. Some authors observed significant cuticular uptake, especially in humid conditions, whereas in dry conditions uptake was primarily stomatal (Livens *et al.* 1992).

Foliar absorption is a rapid process which, in favourable conditions, occurs within a few hours (Levi 1970a). Bukovac *et al.* (1965) studied the translocation speed of radiocesium after foliar absorption in beans, and found that in a few days 85% of the deposited cesium was taken up by the plant. Ludwig (1962) reported that 20-30% of the radiocesium applied to the leaves was immediately absorbed, and that translocation to buds was slower in older plants, whose apical growth was slower as well. The presence of other ions can influence the foliar absorption of radiocesium: increased absorption of radiocesium with increasing potassium concentrations was observed by Levi (1970b); this is probably due to potassium-induced modifications of the permeability of cell walls.

In leafed forests the major initial uptake of radionuclides from an atmospheric source is by the

foliage. Radiocesium is rapidly translocated in the tree and appears in the new foliage growth (Schell & Tobin 1990). However, Withford (1968) found that only c. 4% of the amount of radiocesium applied to the leaves of understory plants in a mesic forest was absorbed and translocated to stem and roots; most of the contamination was transferred to the soil by rainfall. Vallejo *et al.* (1990) studied a Mediterranean forest in Spain after the Chernobyl accident, and found that the leaves which received the deposition from Chernobyl showed radiocesium activity, whereas younger leaves had little or non-measurable, translocated or root uptaken radiocesium. The activity of leaves sprouted after May 1986 had a Cs-134/Cs-137 ratio equivalent to that of the Chernobyl fallout, and the low activity detected in some plots was attributed to retranslocation from older leaves. Block & Pimpl (1990), in a study on mixed forests in Rheinland (Germany), found that the leaves of understory beech trees showed clearly higher activities than those of oak trees; at the time of the Chernobyl peak fallout, beech trees had already sprouted, while oak trees had not; the cesium activities of the leaves decreased only very gradually between 1986 and 1988. Such results, however, could have also been influenced by the fact that in beech the root systems are more superficial than in oaks (Guillitte pers. comm.). Radiocesium is probably stored in the trunk or in the branches during the winter half-year and then transported into the newly formed leaves in the sprouting phase.

Monte *et al.* (1990) studying the behaviour of radiocesium in hazelnuts following the Chernobyl accident, found an exponential decrease of the Cs-137 content of the fruits in a period of three years; evaluation of the Cs-134/Cs-137 ratios in samples of atmospheric particulate, soils and fruits suggests that foliar translocation is the main pathway for radiocesium migration to the fruit, the contribution of root uptake being very small compared with that from foliar uptake. In fact, many authors working on cultivated trees reported an exponential decrease of contamination with time, stressing that contamination by root uptake is a small, if not negligible fraction of total contamination, and that the principal source of contamination of every year's new tree products is the radiocesium inventory in the plant itself. In many cases foliar absorption just after deposition is advocated as the primary source of radiocesium to the tree. As it will be discussed later, however, such statements do not always hold true for

plants of natural ecosystems.

Uptake via roots

According to Tikhomirov & Shcheglov (1994) the process of radionuclide migration and redistribution between the components of forest ecosystems can be divided into two stages. In the first stage, lasting 2-4 years, contamination of plants mainly results from primary aerosol precipitation of radionuclides on the tree canopy. Cesium radionuclides, being capable of incorporation into assimilating organs and of further transfer to other structural parts of trees, are of particular relevance here. The second stage follows the radionuclide transfer from the phytomass into the forest litter and then into the root-inhabited soil layers, and is characterized by a predominance of root uptake. Therefore, the dynamics of radionuclide distribution between the forest components is determined by the difference between two major processes, that of deactivation and that of root absorption. In the initial period, deactivation dominates, with an observed decrease in contamination levels for structural parts of trees. Later, a point of compensation is achieved, followed by a possible enhancement in radionuclide content in the aboveground phytomass, until some quasi-steady state is achieved. In these conditions, the annual transfer of radionuclides from soil exceeds its return with foliage fall only by the content of annual accretion of phytomass. The period for achieving such an equilibrium is of c. 10-15 years (Tykhomirov & Shcheglov 1994).

The principles underlying soil-to-plant transfer of radionuclides are comparable to those pertaining to plant nutrition in general. The relation between soil properties and the soil-to-plant transfer is a complicated matter in which many parameters are involved. Among the most important are: clay fraction, type of clay minerals, organic matter content, pH, soil moisture content and amount of exchangeable ions (see e.g. Nielsen & Strandberg 1988); other factors are related to plant characteristics, such as uptake regulation and mechanisms, and growth stage of the plant. The numerous parameters and the interactions among them make it difficult to determine the effect of each single factor on transfer. Coughtrey & Thorne (1983) in their classical review of the subject, concluded that observed effects are often hardly quantified or explained.

Concentrations of radiocesium in soils are not always a good parameter for calculating reliable

soil-to-plant transfer factors. According to Horrill *et al.* (1990) the concentration of radiocesium in an individual species is related to the radiocesium mobility rather than to total activity in the soil. These authors found higher activities in plants on soils low in mineral matter capable of binding radiocesium. The importance of the soil solution in understanding radiocesium transfer from mineral soil to plant has been underlined by Nisbet & Lembrechts (1990). The soil solution bathes absorbing plant roots, and is the medium from which roots obtain inorganic nutrients (Adams 1974); it provides the chemical environment of plant roots, and defining soil-plant interrelationships in quantitative terms requires a complete and accurate knowledge of soil solution chemistry. Plant availability of an element in soil is expressed by the K_d value. This is defined as concentration per gram of soil, divided by the concentration per ml solution. High K_d values mean a strong adsorption of an ion in soil, and, consequently, a low availability. Reversely, low K_d values indicate high availability. It is now well-known that radiocesium is much more bioavailable in organic than in mineral horizons.

In the following, we shall briefly review the main factors influencing root uptake: pH, clay and organic matter content, the concentrations of other ions, climatic conditions, rooting depths, mycorrhiza, and some plant physiological mechanisms.

Effects of pH on root uptake - Cesium uptake by roots is often claimed to be generally higher in conditions of low pH (Adriano *et al.* 1984, Frissel *et al.* 1990). Studying permanent pastures, Caput *et al.* (1990) found that two years and a half after the Chernobyl accident the transfer factors from soil to plant, calculated on a dry weight basis, were much lower on basic than on acid soil. According to Kerpen (1986, 1988) acidity is even the main factor affecting radiocesium availability in soils, the most acidic soils being those in which radiocesium is the most available. Schuller *et al.* (1988) performed a stepwise multiple regression analysis among transfer factors values and several soil parameters, and found that 67% of the total variance was explained by soil pH. With decreasing pH the specific activities and the transfer coefficients tend to increase; this was explained with the fact that an increase of the H^+ concentration is connected with a decrease of the potassium exchange capacity (Kühn *et al.* 1984). However, other authors (e.g. Fredriksson *et al.* 1966, Bergeijk *et al.* 1992) claim that soil pH has hardly any effect on the soil-to-plant transfer of

radiocesium in the range of pH 3.9-8.4. According to Wauters *et al.* (1994), pH itself has little influence on radiocesium uptake rates, and its role has been overestimated in the previous literature, due to misinterpretation of the results. It appears that pH, being directly related to the Ca-Mg status of the soil, has no direct ion exchange effect upon short-term cesium availability, but is indirectly effective by way of a long-term enhancement of the cesium fixation potential of the soil, which is enhanced by a high (Ca+Mg)/K ratio.

Effects of the organic matter and clay content on root uptake - Not all of the radiocesium present in the soil is directly available to plant roots for uptake (Andersen 1967). The availability of cesium is inversely proportional to the clay content of the soil, being negatively correlated with the adsorption capacity of the clays and positively correlated with the cation exchange capacity of the humus (Cummings *et al.* 1969).

Several studies showed that vegetation growing on organic soils is much more susceptible to systemic contamination by radiocesium than that growing on clay-rich soils (e.g. Sandalls *et al.* 1990, Sandalls & Bennett 1992). This was well-known already from the early years of radioecology (see e.g. Barber 1964, Fredriksson *et al.* 1966, and Marckwordt 1968). The influence of organic matter on the uptake of radiocesium into *Lolium perenne* has been also demonstrated in pot experiments (Barber 1964). Andolina & Guillitte (1990) found a relatively good correlation between the contamination level of plants growing in each station and measurements of the availability of radiocesium in organic horizons. Römmelt *et al.* (1990) measured several understory species, and found that their cesium ratio corresponds well with that of the organic horizons, which indicates that these plants take up cesium mainly from there, although their roots penetrate into the mineral horizon as well.

It is generally claimed that in organic soils, radiocesium is associated with the readily reversible ion exchange complex from which it may be easily desorbed, and remain available for extended periods of time. It is also generally accepted that radionuclides associated with the smaller molecular size fractions will be more bioavailable than those complexed by larger colloidal material. Thus, studies of the speciation of radionuclides in soil solution can provide means for predicting the effects of potential countermeasures (Nisbet & Lembrechts 1990). During the fallout period after the Chernobyl

accident, more than 75% of radiocesium in rainwater from southern Norway was bound to colloids, or particles; most of it was carried by particles with diameters below 1 mm (Devell 1988 a,b, Georgi *et al.* 1988). Haugen *et al.* (1990) studied the availability of radiocesium in the litter and uppermost layer of soil by comparing Chernobyl-derived cesium with Cs-134 in ionic form; they found a relatively low uptake of Chernobyl cesium on the first sampling 1-2 months after the addition of Cs-134 in ionic form to the soil; however, after one year only small differences were evident between these two components. This indicates that plant availability may be largely described as that of cesium introduced in ionic form. According to Bergman (pers. comm.), these observations suggest that the particulate carrier probably influences the behaviour of radiocesium at an early stage after deposition, and might be responsible for the relatively deep penetration observed in sites where deposition was concomitant to intense rainfall. For example, Cremers *et al.* (1990) found that in soils with lower organic matter content radiocesium availability is controlled by the specific sites in the micaceous clay fraction and the ammonium-potassium status of the soil. This well-known fact led several early radioecologists, mainly working with cultivated trees, to assume that in trees root uptake of radiocesium is negligible, since cesium was concentrated in the surface layers of mineral soil, where it was fixed by clays (see e.g. Coleman *et al.* 1963, Johnson *et al.* 1966, Nishita *et al.* 1965). Also recently, this opinion was put forward, and also in this case by researchers working on cultivated trees (Kühn *et al.* 1984, Antonopoulos-Domis *et al.* 1990, 1991, Anguissola & Silva 1992). There is now ample evidence that this does not hold true in most natural forests, where trees, besides being usually in mycorrhizal symbiosis, root in a multi-layered soil, and often have the highest root density in the organic horizons, where cesium is more available for root uptake. Also sandy soils are not capable of binding large quantities of radiocesium (see e.g. Bergeijk *et al.* 1992). Kerpen (1988) and Andolina & Guillitte (1990) found that soil samples with the highest radiocesium availability are those with a high sand content.

As radiocesium deriving from the Chernobyl accident and from weapon tests are distributed differently in soil horizons, their availability to root uptake likewise differs. A greater availability of Chernobyl radiocesium is reported by many authors (e.g. Colgan *et al.* 1990). Some authors ascribe this

difference to the physico-chemical character of the initial carrier, others stress the unequal repartition of "young" and "old" radiocesium in the soil profile as the main reason for the greater availability of the former. Fraiture (1992) claims that in 1989 most of the radiocesium from Chernobyl was still located in the upper part of the holorganic horizons, where it was relatively available for plants, while most of the radiocesium from bombs had already migrated to deeper horizons, where it was strongly adsorbed by clay minerals. Sandalls & Bennet (1992), studying upland grasslands in Cumbria, found that the uptake of Chernobyl radiocesium was up to an order of magnitude greater than that of old radiocesium (i.e. deposited 20-30 years earlier). The rate of uptake of the Chernobyl radiocesium is therefore also expected to decrease markedly over a period of decades. On a shorter timescale, soil-to-plant transfer on clay-rich brown soil fell by about 77% in three years, but on soils with an organic matter content of 88% the decrease was only of c. 17%. Similar results were obtained by Beresford *et al.* (1992) in another study on upland grasslands in Cumbria; initially, after the Chernobyl accident, the transfer of the recently deposited radiocesium was greater than that of aged deposits; four years after the accident, the transfer of Chernobyl radiocesium from the top 4 cm of soil to vegetation was similar to that of the aged radiocesium, because a greater proportion of the deposit migrated further down the soil profile. The movement of Chernobyl radiocesium from organic to mineral layers, and its radioactive decay are the two major factors which contribute to its decline in vegetation. Sombre *et al.* (1994) in an experimental study on Belgian spruce forests, have demonstrated that in the OAh horizons the availability of radiocesium is very weak, in contrast to readily available potassium. According to these authors, it seems that in addition to the clay specific retention, some other more effective retention mechanism exists, which may be related to the simultaneous presence of humus and mineral matters in an intimate mixture. The Oah horizon seems to play the role of a geochemical barrier, which maintains radiocesium in the upper soil levels. As the radiocesium bioavailability in the other upper soil layers is non-negligible, radiocesium transfer to plants which exploit these surface layers is favoured. These results, which are confirmed by those obtained by Nimis *et al.* (1990, 1994) in beech forests of the eastern Alps, could partially explain the negligible decontamination of forests, radiocesium being able to be mobilized by root

absorption, to be carried into the tree and to be partially returned by throughfall and litter to the soil, where it is again available to plants. The reasons for the very weak bioavailability of radiocesium in the OAh horizon, however, are far from being clear. A possible, alternative explanation could be that organic horizons host an intense microbial and fungal life, and that a good share of the total radiocesium could be biologically fixed (see later). When microorganisms die, they liberate radiocesium, which is immediately taken up again by other microorganisms, or by plant roots. This casts some doubts on the relevance of classical soil parameters for explaining the transfer of cesium into plants.

Effects of other ions on root uptake - It is well-known that radionuclide distribution within solid and liquid phases of a soil may be altered by the application of inorganic ions, organic matter or chelates. In particular, the ions NH_4^+ , K^+ , Rb^+ and Cs^+ form a homologous series which shows a greater degree of physico-chemical similarity than any other series of ions, with the exception of halogens. This means that the availability and behaviour of radiocesium in the soil are also dependent on the concentrations of other monovalent cations. These have a double and contrasting effect on cesium uptake; on one hand they facilitate the bioavailability of cesium through processes of cation exchange, on the other hand they concur with cesium for root uptake. In general, higher potassium concentrations will result in less cesium binding to the soil. According to Van Voris *et al.* (1990) the physico-chemical nature of these processes results in differing rates of reaction and equilibrium times for individual monovalent cations, but appears to be governed by the charge and hydrated atomic radius of the interacting ions. Cremers *et al.* (1990) found that for peaty soils with very high organic matter content, radiocesium availability is essentially controlled by the ammonium-potassium status of the soils, and that the bulk of radiocesium is reversibly associated with the ion exchange complex.

It has been recognized for many years that K^+ exerts a depression in radiocesium uptake, presumably as a result of ion uptake competition (see e.g. Menzel, 1954). The application of potassium to soils for reducing radiocesium uptake by plants is one of the countermeasures that has received much attention than most others (Coughtrey & Thorne 1983, Nisbet *et al.* 1990). Grauby *et al.* (1990) demonstrated, in greenhouse experiments,

that in many soils a supply of 5000 kg/ha of K_2SO_4 allows a division of the transfer factor of radiocesium to beans by a factor of ten. Jones *et al.* (1991) found a clear suppression of radiocesium uptake by *Calluna vulgaris* and *Agrostis capillaris* grown in pot culture. Sandalls & Bennett (1992) working on grassland vegetation in Cumbria, found that transfer factors were directly proportional to both exchangeable potassium and cesium, but inversely proportional to the levels of potassium in the soil solution at field capacity. This could be explained by the inability of the plant to differentiate cesium from potassium in the soil water, so that the effect of adding potassium to soil is to reduce the fractional molar concentration of the cesium ion in the total alkali metal pool of the soil solution. In other words, if plants are not thought to differentiate between cesium and potassium, any increase in radiocesium activity in the soil solution could lead to an enhanced uptake of this radionuclide by plants. Conversely, any increase of the potassium concentration in the soil solution could be expected to slow down cesium uptake.

The extent of this processes, however, seems to be variable according to soil type (Nisbet & Lembrechts 1990), and some results of potassium fertilization on radiocesium uptake are rather contradictory. Kerpen (1988) found a very close correlation between radiocesium availability and the K/Cs ratio in soil extracts when adjusted to a parabole. Kirton *et al.* (1990) showed that in upland British pastures radiocesium was transferred to above-ground vegetation as effectively as was potassium during the onset of spring growth. According to Vallejo *et al.* (1990) radiocesium in a Mediterranean forest soil was clearly less mobile than potassium. This is in agreement with the views of Tobler *et al.* (1988), who postulated a predominant low solubility of radiocesium related to their characteristics of the areosols. Clear differences between radiocesium and potassium were also observed by Clint *et al.* (1990) in the release of these elements from decaying litter: in all litters a larger percentage of radiocesium was retained than potassium. According to Shaw & Bell (1991), while it is generally accepted that the roles of the NH_4^+ and K^+ ions are important for the bioavailability of radiocesium in soils, there seems to be some degree of confusion as to the exact effect which each ion can exert. On one hand enhanced K^+ and NH_4^+ concentrations intensify the ionic pressure and make radiocesium more plant-available, on the other hand enhanced K^+ and NH_4^+ levels decrease the Cs-137

uptake by plants. Livens & Loveland (1988) report that a deficiency of bioavailable K and an excess of ammonium will increase the mobility of radiocesium. Schultz (1965) observed that ammonium and potassium fertilization resulted in an increase of cesium uptake by the plants. The exact influence exerted by each ion appears to be the result of a balance between competition for adsorption or absorption by clays and plant roots, respectively (Shaw & Bell 1991). In clay-poor soils the addition of potassium depresses radiocesium uptake by plants (Robinson & Stone 1986), probably because of direct competition for root uptake between Cs and K ions. In the soils with a low cationic exchange capacity, an excess addition of potassium produces an increase of the transfer factors for cesium. Laboratory results by Mitchell *et al.* (1990) demonstrated that rubidium and potassium (at equivalent ionic concentrations) tend to reduce radiocesium concentrations in vegetation; these effects, however, were highly dependent on the particular combination of plant species and soil conditions; field investigations, by the same authors, showed that potassium reduced radiocesium concentrations in vegetation, but only very slightly. Similar results, obtained by Jackson & Nisbet (1990) in soils of upland fell vegetation in Cumbria, highlighted the importance of soil type in determining the effect of potassium on radiocesium and potassium levels in soil solution. In peaty soils potassium applications led to increased potassium concentrations in soil solution without a concomitant increase in radiocesium. These results are in contrast to those for loam and sand, where potassium applications actually increase radiocesium levels in the soil solution, and indicate that Cs-137 is mainly adsorbed in mineral horizons, but mostly biologically fixed in organic soils. Shaw & Bell (1991) maintain that, at least in terms of the absorption mechanism, the behaviour of radiocesium in the environment may be simply a reflection of the behaviour of the normally abundant K^+ and NH_4^+ ions, and that, due to the similarity of the effect of these ions with that of the Cs^+ ion, the transfer factors for radiocesium may vary in a non-linear fashion with the soil solution concentration of these ions. According to Barber (1964), however, the true limiting factor of plant ion uptake from many soils is the diffusive flux of the ion from the soil to the root surface. Hence, in order to fully understand the transfer mechanism from soil to plant of radiocesium, more information is needed on the solution chemistry of the Cs^+ ion, particularly with

regard to the competitive effects of K^+ and NH_4^+ on its sorption-desorption behaviour. Finally, Wauters *et al.* (1994) demonstrated that the $(Ca+Mg)/K$ ratio in soils plays a key role in accelerating the radiocesium fixation process in the specific sites of micaceous clays, which might open some new possibilities in terms of countermeasures.

Effects of climatic conditions on root uptake - The amount of rainfall, light intensity, temperature or air humidity show considerable fluctuations in each growing season, and influence radionuclide uptake, and plant growth and development (Romney *et al.* 1960, Noordijk *et al.* 1992). Furthermore, climatic conditions may affect the availability of a radionuclide in the soil. Variations in climatic conditions have been estimated to cause 3- to 50-fold changes in uptake of radiocesium or -strontium by grass (Squire 1966, Lembrechts *et al.* 1990). According to Noordijk *et al.* (1992) annual fluctuations in transfer factors of radiocesium are an important source of variation and have to be considered when interpreting results of long-term field studies. These annual fluctuations are to a large extent caused by differences in climatic conditions.

Effects of rooting depths on root uptake - Livens *et al.* (1991), studying soil-to-plant transfer in upland areas of Europe, found that radiocesium activities in bulk vegetation reflect those in the rooting zone of the soil profiles. This is an obvious fact, which, however, has been all too often underestimated in the definition of transfer factors from soil to plant, since in natural systems, and especially in forests, it is often difficult and time-consuming to assess average rooting depths of plant species. Furthermore, for calculating transfer factors taking into account rooting depths it is also indispensable to carry out a detailed analysis of radiocesium concentration and behaviour (especially as far as availability is concerned) in the different soil horizons, which is also not an easy task. In natural ecosystems a further complication arises from mycorrhizal symbiosis (see later), which makes almost impossible to exactly estimate the soil portion from which the major uptake of elements occurs. Nimis *et al.* (1990, 1994) found a good correlation between rooting depths and radiocesium concentrations in plant water of almost all vascular species of a natural beech-fir forest of the Carnic Alps (NE Italy), the highest values being reached in plants with shallow root systems, mainly concentrated in the more acid upper part of the

organic layer. Similar results were obtained by Guillitte *et al.* (1994) in a study carried out in a Swedish Boreal forest: deep-rooting species showed far less contamination than species with a superficial rooting system.

Effects of mycorrhiza on root uptake - Most species of vascular plants have evolved to a dependence on mycorrhizae as the most metabolically active parts of their root systems. Mycorrhizal fungi are vital for uptake and accumulation of ions from soil and translocation to hosts because of their high metabolic rates, and strategically diffuse distribution in the upper soil layers; they produce enzymes, auxins, vitamins, cytokinins and other compounds that increase rootlets size and longevity. The fungal mycelium and sporocarps are sources of accumulated nutrients and energy for decomposers and consumers; nutrients and carbon can be translocated from one vascular plant to another by a shared mycorrhizal mycelium (Trappe & Fogel 1977).

Mycorrhiza is one of the least studied, and nevertheless one of the most important factors for understanding the cycling of radionuclides in natural and semi-natural ecosystems. Ecologists working in the laboratory, or with cultivated plants, have often the tendency to underestimate the extent of mycorrhization occurring in natural ecosystems, and its enormous implications in understanding the transfer from soil to plant.

Mycorrhizal associations can be grouped into four classes: those mainly between some higher Basidiomycetes and Ascomycetes with the roots of many forest trees; those between some Basidiomycetes and Orchids, those between a group of fungi and ericaceous plants, and those between fungi formerly assigned to the genus *Endogone* and the roots of many higher plants (e.g. see Harley 1969). Most species of trees have been found to carry mycorrhiza if grown under suitable conditions, and in existing natural forests mycorrhiza is a normal feature of the root systems, whatever the tree. For an optimal development mycorrhiza needs a supply of organic matter in the soil, and thus develops most freely in the organic horizons of the forest soils. It also needs a limited, but not too restricted supply of nutrients, and in fact mycorrhizal formations appear to be the tree's response to low availability of nutrients. Thus mycorrhiza is more common on tree roots growing in a mor or raw humus layer than in a Mull layer. Another important role of mycorrhiza is that of providing water to the

plant; many tree species cannot thrive under water stress without mycorrhizal symbiosis. There are two main types of mycorrhiza: a) ectomycorrhiza, more frequent in trees, in which the fungus develops a compact sheath of hyphae all around the host root, some of which penetrate between the cortical cells, but never inside the cells (Marks & Kozlowski 1973); b) endomycorrhiza, in which the fungus penetrates inside the cortical cells of the root; these can be subdivided in three main groups: a) vesicular-arbuscular mycorrhiza, which is by far the most widespread, b) mycorrhiza of the *Ericales* (see later), and, c) mycorrhiza of Orchids (Sanders *et al.* 1975). Ectotrophic mycorrhiza is more common in forest trees. However, vesicular-arbuscular endotrophic mycorrhiza is far more widespread among plants, and the roots of almost every plant species growing on some soils appear to be mycorrhizal.

Data by Grant *et al.* (1985), Bunzl & Kracke (1986), Horrill *et al.* (1990), and Strandberg (1994) show that ericaceous species often exhibit a high uptake of radiocesium. Several British authors (e.g. Colgan *et al.* 1990) reported particularly high radiocesium activities in *Calluna vulgaris*, a common heathland plant of the *Ericaceae*. In general, the roots of these plants basically grow in the lower part of the organic horizons, where cesium availability is high, and, above all they have a particular type of mycorrhiza, called ericoid mycorrhiza (Strullu 1985). This is produced by the Ascomycete *Pezizella ericae*, and perhaps also by some Basidiomycetes of the genus *Clavaria*. The fungus can account for up to 80% of the total volume of the root, with an average of c. 43%. This means that in many *Ericales* the root biomass is actually constituted, to a large extent, by a fungus. The importance of mycorrhiza for explaining radiocesium levels in higher plants has been underlined by Guillitte *et al.* (1990c, 1994) and Wirth *et al.* (1994). Fungal mycelia cover a much larger soil area than that covered by plant roots. In estimating transfer factors from soil to plant, soil samples are usually taken in the rooting zone of a plant. But, if fungi contribute significantly to the nutrient supply of their hosts, the question arises how representative soil samples taken directly from the rooting zone might be (Wirth *et al.* 1994).

There is evidence that complex nutritional phenomena occur between the fungus and the root (Bowen & Smith 1981). It has been demonstrated that the fungus, when grown in pure culture, is able to utilize phosphorus both in the organic and the

inorganic forms, contrary to what happens for the roots of higher plants (Gianinazzi-Pearson & Gianinazzi 1980). According to Colpaert & van Assche (1988) and Colpaert (1991) different species of ectomycorrhizae have different behaviour as far as the absorption and transfer of heavy metals to the host plants are concerned; in some cases the fungus induces higher concentrations of metals inside the plant, in other cases the outer mycorrhizal sheath acts as a barrier, preventing the transfer of metals to the plant.

In a literature survey Leising (1986) came to the conclusion that cesium behaves like potassium within the plant, but during root uptake it is always discriminated with respect to potassium. Similar results were obtained by Nimis *et al.* (1994) and Wirth *et al.* (1994). It is not unlikely that cesium is discriminated a second time during the transfer from the mycorrhizal mycelium into the root cells of the host, in which case fungi would accumulate more cesium the more intensively they supply their hosts with nutrients (Wirth *et al.* 1994). According to Olsen *et al.* (1990), if plant roots have a higher preference for potassium versus cesium than the mycobiont, they would act as a relatively selective potassium sink in the soil, which would increase the Cs/K ratio of the cations available for the mycobiont; it seems likely that even small differences in selectivity between plant and mycobiont may have a large impact on the cesium level in both partners of the mycorrhizal symbiosis. Rogers & William (1986) demonstrated the favourable influence of another type of vesicular-arbuscular mycorrhiza on uptake of radiocesium in *Melilotus officinalis*. The influence of vesicular-arbuscular mycorrhizae on the uptake of radiostrontium by soybeans was studied Jackson *et al.* (1973).

Given the importance of mycorrhiza in natural and semi-natural ecosystems, more detailed studies on the role of mycorrhiza in radiocesium uptake are badly needed.

Physiological mechanisms of absorption and transport - Plant roots have a remarkable ability to take up several ions from solution concentrations much lower than those in the plants. Earlier studies claimed that radiocesium is taken up by roots as easily as potassium (e.g. Collander 1941). As early as 1952, Epstein & Hagen suggested that, in roots of barley, K^+ , Rb^+ and Cs^+ enter into the cells through the same absorption mechanism. For this reason, Rb_{86} has been often used as a substitute of K in

studies of ion uptake by roots. More recent physiological researches (e.g. Sacchi & Cocucci 1991, 1992, Sheahan *et al.* 1993) showed that the absorption kinetics of the three ions are competitive with each other, a hypothesis corroborated also by the fact that cesium is accumulated by the systems transporting potassium in microbial cells (Bossemayer *et al.* 1989) and in the alga *Chlorella salina* (Avery *et al.* 1993). According to Coughtrey *et al.* (1990), continued uptake of radiocesium by selected plant species could reflect the low potassium status of the soils involved. However, these authors found that potassium status was not an universal explanation for the different patterns in cesium concentrations among species. There is evidence that the chemical behaviours of cesium and potassium are very similar, but not completely identical (e.g. Nishita *et al.* 1965, Wallace 1968), and now it is well established that cesium is generally discriminated versus potassium during root uptake and in the penetration into plant cells. It should be underlined that Cs^+ reduces the electrical conductivity of animal cell membranes, and therefore is likely to block the potassium channels (Clay & Schlesinger 1983). Recent, similar results found in the plant *Chara contorta* suggest that this effect does also occur in plant cells (Zanello & Berrantes 1992). The apparent contradiction between the inhibitory action of cesium on potassium transport and potassium absorption is a consequence of many complex biochemical mechanisms which are involved in potassium transport, and mainly to the different affinity of Cs and K to transport molecules.

Potassium can be absorbed by plant roots from a wide range of concentrations (from milli- to micromolar). There is evidence that the absorption of the cation includes more than one kinetic (Epstein *et al.* 1963), and it is generally accepted that more than one transport system is found on the plasmalemma (Kochian & Lucas 1982). The presence of different transport systems is also suggested by results obtained by utilizing inhibitors with different specific sensitivities to the different kinetic systems (Kochian & Lucas 1982, Kochian *et al.* 1985). The absorption of potassium seems to be mediated by a high affinity system which is predominant at low concentrations, which is made possible by an energized carrier, while at high concentrations potassium transport could occur through channels. According to Cocucci (com. pers.) laboratory experiments in maize roots show substantial selectivity differences between Cs^+ and

K^+ both in systems operating at high and in those operating at low concentrations. In natural soils the relative concentrations of potassium and cesium are often quite different; whereas potassium is often present at high concentrations in a wide range of natural soils, the relative concentrations of cesium are of several orders of magnitude smaller.

The potassium concentration inside a plant cell is restricted into narrow ranges, otherwise the cell metabolism would not work. This means that potassium uptake is practically independent from soil parameters. If the available K-concentration in soil is too low, the plant will grow more slowly, but the K-concentration in the cells will be not significantly different from a well nutrient-supported plant. Furthermore, similar ratios of Cs-137/K in different organs of the same plant indicate that in plants cesium behaves very similar to potassium, and suggests that radiocesium activity in plants is regulated via the potassium metabolism. Steffens *et al.* (1980) observed in three agricultural plant species, grown on 14 different soil types, a variation of several orders of magnitude in the radiocesium contents. By standardizing radiocesium concentrations on average potassium levels, however, the data showed a much smaller variation. This fact suggests that a certain ratio of plant available Cs-137/K in soils is reflected in the ratio found in plants. These results are valid assuming a homogeneous distribution of radiocesium in the soil. They are difficult to verify in the case of undisturbed soils, as the ratio of the plant available Cs-137/K differs in the different layers. Furthermore, the different rooting depths of the plants, the degree of biological fixation of radiocesium in the soil, the influence of mycorrhiza on plant uptake complicate very much the system and enhance the uncertainties of prognostic estimations.

A peculiar case is that of aquatic plants. Nelin & Nylén (1994) found much higher concentrations of radiocesium in *Nymphaea* than in other terrestrial species of a Boreal forest ecosystem; these authors explained this fact stating that aquatic plants are known to concentrate mineral salts, due to the special osmotic stress they are exposed to; therefore, it would be reasonable to expect high concentrations of cesium in such plants. However, higher concentrations of salts are common in halophytic aquatic plants, or in desert and semi-desert plants, and not in plants growing in fresh water. A more plausible explanation of the high radiocesium concentrations found in *Nymphaea* could be the fact that the data were expressed in Bq/kg of dry weight,

and that the water content of this species is much higher than that of all other measured terrestrial species (see later).

Translocation inside the plant

Radiocesium translocation within plants was first studied experimentally by artificial introduction of Cs-137 into the body of trees or other cultivated plants (Olson 1965, Waller & Olson 1967, Hoffmann 1972). Bukovac *et al.* (1965) studied the translocation speed of radiocesium after foliar absorption in beans, and found that in a few days 85% of the deposited cesium was uptaken by the plant; about the half was transferred to other parts of the plant, and even the roots showed considerable concentrations. Middleton (1959) found that in potatoes c. 50% of the radiocesium absorbed by leaves was translocated in the buds. Witherspoon (1962, 1964) found that in white oak trees Cs-134 was present in all parts of the crown 1.5 hrs. after trunk inoculation; downward movement in the trunk was slower than upward movement. Aarkrog (1975) applied radiocesium to the leaves of corn, and found that relevant translocation to the seeds was detectable also when the application occurred at a very early stage of the plant life. The high translocation speed of radiocesium is discussed in a large body of literature; for a review see Coughtrey & Thorne (1983). These results indicate that, once inside the plant, radiocesium, like potassium, is highly mobile. There is evidence that cesium and potassium behave similarly inside the plants. Potassium is employed as a counter ion, and is important in a wide variety of plant processes including sugar transport, enzyme function, osmotic balance, and charge neutralization for structural macromolecules. Wirth *et al.* (1994) showed that the absolute activity of radiocesium in stems, leaves and fruits is variable, but the K/Cs ratio in different organs is about the same, indicating that the distribution of radiocesium within a plant follows the K⁺ pattern. Nimis *et al.* (1994) studied the K/Cs ratio between leaves and stems of several vascular plants of a natural mixed forest in the Carnic Alps (NE Italy), in order to test whether, once inside the plant, the two elements have a similar behaviour. The results were quite clear, and quite surprising. There is a strong difference between ferns (and, in a lesser degree, of geophytes) and all other plants. In ferns the concentrations of cesium in the leaves are always higher than expected, whereas the ratio is

more or less constant in the other vascular plants. This fact still awaits a physiological explanation. The most probable hypothesis is that ferns and vascular plants differ in some important transfer mechanism concerning cesium and potassium.

Radioactive fallout due to the Chernobyl accident provided the opportunity to carry out translocation studies under real conditions. In a spruce stand, needles constitute the main reservoir of potassium (44%), trunk (wood and bark) comes second (34%) and branch wood third (22%) (Nys *et al.* 1983). If the distributions of cesium and potassium follow the same patterns in trees, one has to conclude that leaves, short after deposition, are the principal reservoir of radiocesium, and hence the main source of transfer of radioactivity from trees to soil in coniferous forests. Block & Pimpl (1990) report that in Germany spruce needles sprouted before 1985, measured in 1986, showed the highest contents of radiocesium. However, already in the autumn of 1987 it was no longer possible to distinguish the cesium activities of the different needle years. A similar situation was reported by Tobler *et al.* (1988). This indicates that radiocesium was easily transported from older parts of the plants to new shoots. Cesium concentrations in current needles of Norway spruce have been observed to be higher than in older needles (Cousen 1989, 1991); other authors, however, such as Bergman *et al.* (1988) and Nylén & Ericsson (1989), observed that radiocesium concentrations in the current needles of 45-years-old Swedish Scots pines were lower than in the older needles. Raitio & Rantavaara (1994), studying the concentrations of radiocesium in Scots Pine needles in southern Finland, found that in 1987 the concentrations in 2-year old needles were higher than those in the current needles, whereas in the following years the situation was the opposite; they attributed this fact to resuspension phenomena; the increasing concentration of radiocesium in the current needles was attributed, at least in part, to the radiocesium stored in the bark of trees during fallout being gradually translocated into the phloem, and from there to emerging needles.

Variation in cesium concentrations among different parts of the plant is of considerable significance in respect of transfer to grazing animals. There is evidence that internal translocation, at least in the early period after deposition, plays a relatively minor role in radiocesium transfer within the ecosystem, compared to the transfer effected by rainfall (Bergman pers. comm.).

Seasonal phenomena

Another potential source of variability in radiocesium concentrations in plants are the fluctuations of radionuclides in plant tissues during the growing season. Great variations during the year were measured by several authors (e.g. Bunzl & Kracke 1988, Salt & Meyes 1990, Colgan *et al.* 1990, Sandalls & Bennett 1992). If seasonal variations in cesium uptake among plant species do exist, this fact must be taken into account by radioecologists, and by those interested in evaluating the radiocesium content of the diet of grazing animals. Fluctuations may have different causes, which will be briefly discussed below.

Growth dilution - A factor which may determine a progressive lowering of radiocesium in plants is the so called "growth dilution". The mass concentration of any material associated with the vegetation will decrease at about the same rate as the plant is growing (Miller & Hoffmann 1983). Growth dilution phenomena were observed by Monte *et al.* (1990) in several fruit-trees. Eriksson (1991) studying grass, barley and peas in Sweden after the initial interception during growth, found that the apparent residence half-time of radiocesium in grass was about one month, and that the reduction in nuclide content up to the harvest time mainly depended on dilution by growth. Eriksson & Rosén (1991) found that dilution played an important role in reducing the radiocesium concentrations of grass during the first year after deposition, with most of the radioactivity having been intercepted by plants. However, it should be stressed that dilution phenomena are of importance only when plants are contaminated by direct deposition, without significant root uptake. According to Anderson (1973) the main factor limiting the supply of potassium (and hence, probably, also of cesium) to the growing plant shoot is not concentration, but some intrinsic factor connected with plant growth. In some cases it is possible to show correlations between relative growth rates and rates of transport to the shoot and it seems that some control system operates from shoot to root regulating the import to the growing shoot. This means that in a growing plant "dilution" phenomena may be counterbalanced by an input of new radiocesium from the roots, influenced by growth itself. Perhaps it is not a chance that most "dilution" phenomena were reported from cultivated trees, growing on soils rich in clay, where cesium availability, and hence root

uptake, are generally low (e.g. see Antonopoulos-Domis *et al.* 1991).

Weather conditions - Another factor which might determine seasonal fluctuations in plants is related to weather conditions, especially rainfall and temperature. Salt & Meyes (1990), studying hill pasture vegetation in the United Kingdom, found that all measured species showed very similar and rather pronounced fluctuations in their radiocesium contents between June and October. In all green fractions a first peak occurred at the beginning of July, followed by a second peak in mid-August, after which levels continuously declined until the end of the grazing season. An indication that uptake of radiocesium by many plant species may show temporal variations during the growing season was also found by Colgan *et al.* (1990) in their study of Irish pastures. Lembrechts *et al.* (1990) demonstrated that weather conditions affect both active uptake and external contamination, causing fluctuations in the concentration of radionuclides in plants, which mask the effect of their progressive fixation by the soil. Seasonal variation in cesium levels of upland grassland in Cumbria was observed by Sandalls & Bennett (1992), who attributed it to climatic conditions, although they were not able to identify the factors responsible. This seasonal pattern was very similar to that observed by Brasher & Perkins (1978) for potassium levels in grasslands on the Pennines in 1969.

Plant growth and ageing - Another source of seasonal variation is related to plant growth and ageing: Coughtrey *et al.* (1990), working in British pastures, showed that in some species (e.g. *Luzula sylvatica*) cesium is stored in the basal parts of the plant during the winter months, and that levels in senescent leaves are lower than in green leaves. Some species of ferns, studied by Henrich *et al.* (1990) show a much larger contamination in older plants which were affected by the original Chernobyl fallout. In these, a much larger amount of radiocesium is available for translocation to the rhizome in autumn with subsequent withering of the fronds and relocation to the new frond in spring. In younger individuals, on the contrary, the basic contamination is caused by soil-to-plant transfer only. Coughtrey *et al.* (1990) developed a model to describe radiocesium transfer in British upland pastures, based on a comprehensive review on potassium distribution and dynamics in upland ecosystems. Direct data suggested that some species

might show distinct seasonal patterns in radiocontamination; the model provided an adequate representation of the transfer to live shoots that occurs during spring and the subsequent decline that occurs during autumn and winter. The radiocesium specific activities of several plants of Boreal ecosystems, measured in Sweden by Bergman *et al.* (1991), decreased rapidly during the early summer of 1986. This very fast decline was attributed to a combination of removal during rainfall, translocation within the plant and dilution in growing parts of the plants. After this initial decline, the changes in concentration of Cs-137 are generally much smaller, and the authors assumed that physical decay will generally govern the change in the content of radiocesium in the ecosystem. Sombre *et al.* (1994) have demonstrated that in forest trees cesium and potassium have a similar behaviour, characterized by distinct seasonal patterns; radiocesium seems to follow potassium in throughfall water; moreover, similar acropetal and basipetal translocation phenomena occur in relation to the season. Duvigneaud (1964) had already shown that the potassium pool increases in spring, stabilizes in summer and decreases in Autumn. This autumnal decrease in the potassium concentration in leaves is attributed partly to the washing off and partly to the translocation to trunk, branches, roots, etc. On the other hand, Ranger & Bonneau (1984), observed that 74% of the potassium necessary to the biomass production comes from the older parts of the tree, such as preexisting wood and old needles. An opposite movement is observed at the end of the summer and in autumn, when foliar potassium is redistributed into the woody part of the tree. During this period of decreasing cell activity, cellular potassium leaves the cell (Mengel & Kirby 1987) and then may be more easily leached from the leaves.

A main problem for interpreting the previous literature on seasonal variation of plant radioactivity is the fact that most data are expressed on a dry weight basis. It could be that several of the reported "fluctuations" were not due to variations in radiocesium concentrations in plant water, but to the seasonal variation of plant texture, due to ageing phenomena (see later).

Seasonal rhythms of mycorrhization - Bolognini & Nimis (1995) measured the radiocesium activities of several plants of a mixed forest stand in the Carnic Alps at intervals of about 30 days for two years, expressing the activity on a water basis. They found that many species showed distinct, regular and

often very pronounced seasonal rhythms in radiocesium activity. At least for a species of fern, it was possible to demonstrate a direct correlation between the temporal pattern of radiocontamination and the seasonal development of endomycorrhiza. The highest concentrations of radiocesium were reached in correspondence to the maximum, the lowest to the minimum development of the mycorrhizal infection, which, in this species, has a constant seasonal rhythm. Considering the high number of species showing distinct seasonal patterns found by Bolognini & Nimis (1995) it is very probable that in natural ecosystems, and especially in forests, mycorrhiza-induced seasonal fluctuations are more the rule than the exception. It follows that transfer factors for the same species growing on the same soil may exhibit a very large variation during the relatively short time of a growing season. This has an important consequence for studies on long-term trends in radiocesium concentrations in plants of natural ecosystems, as great care should be made of sampling the plants in the same period of the year.

The expression of radiocontamination in plants

Until now, for explaining the high variability of radiocesium specific activities found in plants of natural ecosystems, we have taken into consideration some main eco- and physiological factors influencing radiocesium uptake by plants. There is, however, another, consistent source of variability, which, being a methodological matter, has nothing to do with these factors, i.e. the expression of radiocontamination of plant material. Due to the strong historical links of radioecology with radiation protection, in many earlier studies radiocesium activities in plants (including mushrooms) were expressed on a fresh weight basis. As, however, the water content of plants is subject to short-term fluctuations (i.e. after a rainy period), nowadays most authors use the conventional expression on a dry weight basis.

It is well-known that most of the potassium present in the plants is free in solution inside the cell, or in the apoplastic water. Due to its chemical similarity, this is probably true also for cesium. The plant cell differs from the animal cell in having a cell wall, whose thickness is extremely variable, even among organs of the same individual. Depending on the thickening of cell walls, the water content of plant material, compared with its dry weight, is very variable, both among and within species. Even in a well-buffered environment, such

as a beech forest, the water content of different species may range from 50 to 90% (Nimis *et al.* 1988), and a still higher range is reached if different parts of the same species are considered (e.g. roots versus fruits). The interpretation of the role of potassium in the plant cell has been often complicated by the conventional expression of the data on a dry weight basis. A physiologically much more relevant expression is that obtained calculating potassium concentrations on the basis of the water content of the plant, or of the tissue under study (Pitman 1975, Leigh & Johnston 1983a, b, Leigh & Wyn Jones 1984). This expression has clarified many contradictory results obtained in the past; for example it has been seen that, contrary to previous results, the concentrations of potassium are more or less stable during the growth of the plants (Leigh & Johnston 1983a,b); critical concentrations calculated on a water basis are not age-dependent, as those calculated on a dry weight basis, which tend to decrease with age (Ulrich & Hills 1967).

An analogous problem exists in radioecology. The dry weight of plant material is not necessarily correlated with the quantity of plant water; this means that the expression of radiocontamination on a dry weight basis does not reflect with accuracy the actual concentration of radiocesium inside the living parts of plants. This may be of little relevance in radiation protection, but becomes a consistent source of error in radioecology, especially when the following problems are addressed:

- 1) comparative studies of transfer factors among different species;
- 2) translocation of radiocesium in different parts of the same plant;
- 3) study of seasonal fluctuations during the growing season.

Nimis *et al.* (1988) have discussed this problem, suggesting that, as in the case of potassium, it could be overcome by expressing the radiocesium activities in plants on a water basis, i.e. on the difference between fresh and dry weight. Nimis *et al.* (1990, 1994), studying many species of vascular plants in a complex forest ecosystem in the Carnic Alps, demonstrated that the expression of radiocesium in Bq/l resulted in a much lower degree of infra- and intraspecific variability, and that this allowed to find significant relations with soil features and with the rooting depths of the different plants. Some of the main errors of interpretation due to the use of the current expression of radiocesium activities in Bq/kg dry weight were illustrated by Bolognini & Nimis (1995). They showed that the

traditional expression on a dry weight basis could easily bring to false interpretations as far as transfer factors soil-to-plant, translocation rates within plants, and seasonal fluctuations in leaves were concerned. For example, young leaves generally contain much more water than senescent leaves; at equal concentrations of radiocesium in the leaf cells, the expression of radioactivity on a dry weight basis would suggest that in younger leaves radiocesium concentrations are higher than in old leaves, or even that during plant growth there is a translocation of radiocesium from old to new leaves. Most probably, the decline in radiocesium concentrations from young to old leaves reported many times in the literature was actually due to trivial and well-known ageing processes which have nothing to do with radiocesium behaviour within the plant.

A critical re-evaluation of the results of many radioecological studies on the light of these considerations has still to be attempted.

Transfer Factors

The current approach to the soil-plant transfer of radionuclides is based on the measurement of Transfer Factors (TF). The soil-to-plant TF is usually defined as a concentration ratio between the specific activity in plants and that in the soil. Its relative ease of measurement has led to the production of a wealth of data, with several efforts aiming at correlating TF values with various soil properties. Kühn *et al.* (1984), for example, measured the specific activity of radiocesium of soil and grass in many localities of NE Germany to calculate TF values; they found that TFs are a function of several soil parameters, and mainly of soil pH. The TF model implies that plant and soil concentrations are linearly related and that the relationship has a zero intercept (Sheppard & Sheppard 1985).

The problem with TF values soil-plant is their extremely high variability, especially in natural ecosystems. The reported values of some parameters may range over one or two (Miller & Hoffmann 1983), or even four (Boikat *et al.* 1985) orders of magnitude. According to Sheppard & Evenden (1990) the available data most often do not support the assumptions inherent in the use of TF to represent plant uptake. The use of simple TF has the following main limitations: 1) it uses total activity concentrations in soils, and does not consider speciation and bioavailability; 2) the TF is defined for steady state conditions, and therefore it takes no account of the dynamics of the process; overall

radionuclide concentrations in a plant may be subjected to changes during its growth phase, especially if the data are expressed in Bq/kg dry weight; 3) TF assumes linearity between soil and plant activity concentrations (Nisbet & Lembrechts 1990). Haunold *et al.* (1986) tried to summarize transfer factors for several radionuclides, with values conventionally referred to fresh weight of cultivated plants; these authors underlined the fact that if rooting depth is not taken into consideration, due to the unequal distribution of radiocesium in the soil profiles, the calculated TF values might be totally unrealistic. According to Sweeck *et al.* (1990) it appears that the tools for making TF predictions on the basis of soil properties are quite limited at this stage. The difficulty stems from the fact that a TF is influenced by soil and plant properties, as it has been discussed before.

To reduce the uncertainty, transfer equations were developed which tried to respect parameters that are of influence on uptake rates, and attempts were sometimes made to relate the ion uptake of plants to relevant soil parameters by performing correlation analyses between those parameters and ion uptake. On the other hand, quite often these parameters could not be fit into sound mathematical relations with respect to the uptake of radionuclides by plants. Schaller *et al.* (1990) could demonstrate that, considering the potassium concentration in agricultural plants, the uncertainty of the radiocesium calculations in plants is significantly reduced. Desmet *et al.* (1991) demonstrated that the equation can be further improved by taking into account only the Cs-137 activity in the soil solution, instead of the total activity.

In complex systems, a more detailed knowledge of the ecology of the different plant species may be of great help in selecting those factors which might be more relevant in influencing root uptake. The different horizons of undisturbed soils show different soil characteristics, and it is difficult to analyze from which horizon different plants take up most of their nutrients. Furthermore, as a relevant share of the radiocesium is biologically retained, especially in the organic horizons, this radiocesium behaves differently than free ions in soil. A further source of uncertainty is the influence of mycorrhiza on the uptake rates of higher plants. Due to this complex situation it might be that simple transfer factors, if well-calculated, are more reliable than transfer equations in describing the uptake of radionuclides in natural systems, because the values for the

parameters of interest are not available, or vary within large ranges.

The previous considerations could suggest that the factors involved in soil-to-plant transfer are so many, and so complicated, that any attempt towards the construction of reliable mathematical models is likely to fail. However, there are examples, such as the study by Kühn *et al.* (1984) which demonstrate that if most of the really relevant parameters are considered, one can find correlations which can be integrated into mathematical models by finding an analytical form of the functional relationships among parameters. Our impression is that simple TFs, as those which are still in current use, are, in the present stage of development of radioecology, more detrimental than useful for the progress of this science, although they certainly can play an important role in radiation protection studies. They mask the complexity of the systems under investigation, and their obviously high variability hampers any serious attempt to develop really predictive and reliable models of radiocesium cycling in the ecosystems. The fact that transfer factors are easy to calculate and are readily understandable by laymen is not a good argument in favour of their extended use in radioecological studies, as such simple and rough tools are not always adequate for studying the dynamics of complex polycyclic systems.

CONCLUSION

In the radioecological literature there are several studies on cultivated plants and agricultural ecosystems, while much less attention was devoted to wild plants and natural ecosystems. Among the latter, grasslands were more extensively studied, both because of their relatively simple structure and of their economic importance as a source of forage. Forest communities and forest plants, on the contrary, have been less studied, and constitute a particularly difficult and complex subject. In complex natural ecosystems several species avoid root competition by exploiting different soil layers with different biological, chemical and physical characteristics, many plants are in mycorrhizal symbiosis with fungal species, the chemical properties of the soil horizons pertaining to the same soil type may be very different, influencing the uptake rates. Furthermore, microgeomorphological variation may induce sharp micropatterns in radiocesium concentrations in the soils. Thus,

spatially contiguous species might have their roots in widely different soil types, or in different horizons of the same soil. Finally, the current expression of radiocontamination in plants on a dry weight basis was the source of a substantial degree of error in the interpretation of radioecological data. Therefore, it is not surprising that radioecological data concerning plants of natural ecosystems are characterized by high dispersion, and that reliable generalizations seem often impossible.

A large number of models concerning the environmental transfer of radionuclides have been developed thus far, and a great deal of information on parameter values has been collected during decades of research. Nevertheless, environmental transfer parameter values, and, consequently, the results of models, still show a high degree of uncertainty (Monte 1990). Increased research on natural and semi-natural ecosystems after the Chernobyl accident has shown a still higher variability of radioecological data concerning these systems, as compared with the much simpler agricultural systems which were the main object of study during the early years of radioecology. The previous considerations might clarify the main reasons for the failure in developing reliable predictive models, and this is still more understandable considering that often generalizations were sought on the basis of such rough parameters as the transfer factors discussed in the previous paragraph, which totally disregard the complexity of natural ecosystem.

However, there is evidence that, by using a sampling strategy which takes into account the complex compartmentation of natural ecosystems, it is actually possible to find good correlations between soil parameters and radioactivity concentrations in plants.

As far as the future developments of radioecology are concerned, my personal opinion is that there are two main, important trends, one leading towards ecophysiology, the other towards general system ecology (Palo 1991). Ecophysiological studies are needed in order to acquire an indispensable basic knowledge on the mechanisms involved in cesium absorption by leaves, roots, and mycorrhiza, and on those concerning the transport and the physiological role of cesium and potassium within the plant. On the other hand, several data already exist on the cycling of nutrients in natural ecosystems, and the results of radioecology should be incorporated in the existing models much more thoroughly than it has been done

until now. In particular, more attention should be devoted to generalization of the results, as many radioecological studies do not go outside the scope of local, or episodic studies. This can be best accomplished through a closer link between radioecology and plant ecology, and especially vegetation science. According to Horrill (1983, 1990) studies on radionuclide deposition indicate that there are significant relationships between vegetation type and the amount of contamination held by the vegetation and its associated soils (see also Kovar 1990). Unfortunately, much of the work performed on the Chernobyl fallout has rarely incorporated exhaustive vegetation data, and has relied mainly on describing the vegetation in terms of the dominant species. It is often the case that what at first appears to be a homogeneous area is in reality a mosaic of two or more vegetation types. The question of scale then becomes important as sampling at different sizes can produce totally different results. By examining the wide range of plant communities and their associated soils throughout Europe, it should be possible to isolate the main controlling factors, and to reach a higher degree of generalization.

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ANALISI FITOSOCIOLOGICA E VALUTAZIONE AGRONOMICA DI PASCOLI SUBALPINI (MALGA PADEON, CORTINA D'AMPEZZO, NE ITALIA)

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Keywords: Abandonment, Agronomic value, Calcareous pastures, Subalpine vegetation, Recolonization.

Abstract: PHYTOSOCIOLOGICAL ANALYSIS AND AGRONOMIC EVALUATION OF SUBALPINE PASTURES (MALGA PADEON, CORTINA D'AMPEZZO, NE ITALY). In this study the vegetation and the agronomic evaluation of subalpine pastures are described. These pastures are developed from 1800 to 2100 m on calcareous soils and have not been used for grazing since the early '80s. The following vegetation types have been identified: *Poa alpina* pasture (*Poion alpinae*), *Carex davalliana* fen (*Caricetum davallianae*), *Carex ferruginea* pasture (*Caricetum ferrugineae trifolietosum*), *Sesleria albicans* pasture (*Carici ornithopodae-Seslerietum albicantis*), calcareous scree community (*Thlaspietea rotundifolii*), *Pinus mugo* scrub (*Erico-Rhododendretum hirsuti*) and *Pinus cembra* wood (*Vaccinio-Pinetum cembrae rhododendretosum hirsuti*). The ecological evaluation of the vegetation types was based on an indirect analysis by means of the Landolt indices. Some dynamic trends have been hypothesized on the basis of the ecological characterization. The indirect agronomic evaluation of the pasture types was based on the goodness values (Gütenzahl) of each species.

Introduzione

Nell'ambito delle attività tradizionalmente esercitate nella regione alpina italiana l'utilizzo dei pascoli di alta quota ha avuto per lungo tempo un ruolo importante non solo sotto l'aspetto strettamente zootecnico ma anche nei riguardi della salvaguardia ambientale.

L'importanza di queste colture ha destato sempre un notevole interesse nel mondo scientifico, che ha prodotto in tempi recenti numerosissimi studi volti a conoscere gli aspetti vegetazionali, meno frequentemente studi relativi all'aspetto pabulare (Lausi *et al.* 1981, Bezzi *et al.* 1984, Bezzi 1985, Grignani *et al.* 1990) e alla dinamica vegetazionale conseguente all'abbandono o alla ripresa del pascolamento (Feoli *et al.* 1980, Acutis *et al.* 1989, Tappeiner & Cernusca 1991, Ziliotto & Scotton 1993).

Con riferimento a tali problematiche il Dipartimento di Agronomia dell'Università di Padova ha partecipato ad "Integralp", un programma di ricerca europeo finanziato dalla C.E.E. e riguardante l'evoluzione dei sistemi silvo-pastorali.

Nel quadro di tale programma il Dipartimento

citato ha eseguito lo studio dei pascoli di Padeon, malga abbandonata da circa un decennio, con l'obiettivo di descriverne le caratteristiche fitosociologiche, il valore agronomico e di analizzare la dinamica vegetazionale determinatasi in conseguenza della cessazione del pascolamento.

Area di studio

Malga Padeon (Cortina d'Ampezzo, Belluno) è posta nel bacino del fiume Piave ed è interamente compresa nel Parco naturale delle Dolomiti d'Ampezzo (Fig. 1). Si estende su circa 225 ha esposti prevalentemente a SO e situati tra i 1790 e i 2100 m s.l.m.; 40 ha sono occupati da pascolo vero e proprio e i restanti sono costituiti da boschi, arbusteti, rocce e ghiaioni.

I massicci circostanti Padeon sono costituiti da rocce sedimentarie di varia natura (argille, marne, calcari, dolomie, arenarie e gesso) dal cui disfacimento si è originato il substrato pedogenetico della malga, formato da accumuli detritici a chimismo eterogeneo.

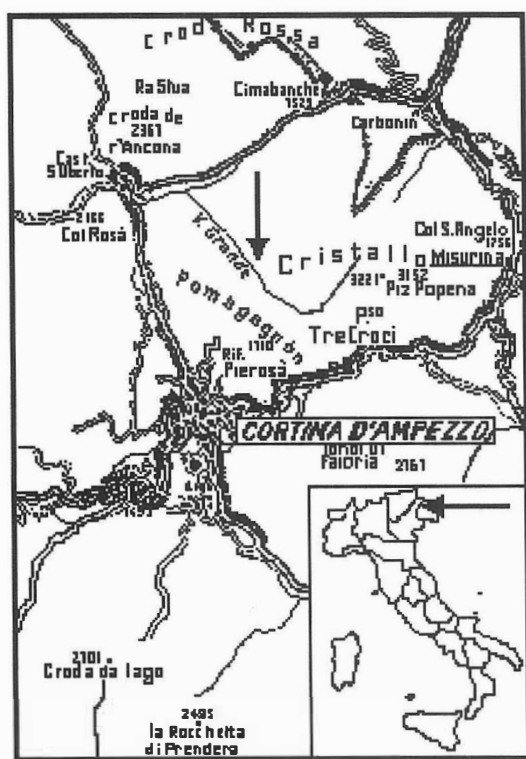


Fig. 1 - Localizzazione geografica di Malga Padeon (freccia).
Geographical location of Malga Padeon (arrow).

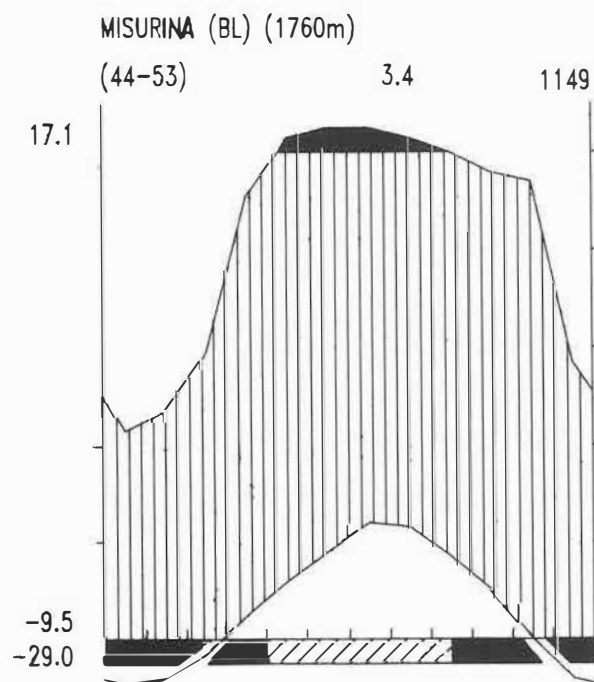


Fig. 2 - Climadiagramma di Misurina secondo Walter & Lieth (1960).
Climate diagram of Misurina according to Walter & Lieth (1960)

Secondo la nomenclatura della Carta dei suoli d'Italia (Mancini 1966) i terreni di Padeon rientrano nell'associazione "Rendzina, suoli bruni calcarei e litosuoli", tipologie effettivamente osservate su vaste superfici dell'area di studio assieme a pseudogley e suoli bruni lisciviati rinvenute su aree meno estese.

Sulla base del climadiagramma della vicina stazione di Misurina (Fig. 2), costruito in base a dati rilevati nel periodo 1926-1969 dal Ministero dei Lavori Pubblici, il clima dell'area in esame è caratterizzato da una temperatura media annua di 3.4° C e da un massimo estivo delle precipitazioni. Le precipitazioni nevose iniziano normalmente in dicembre e l'innevamento può permanere fino all'inizio di giugno.

Sulla base della carta dei complessi di vegetazione di Cortina d'Ampezzo (Pignatti 1981a), le tipologie forestali, presenti nell'area in esame fino ai 2100 m s.l.m., sono riconducibili alla pecceta subalpina tipica, alla cembreta e alla mugheta. Le formazioni erbacee, diffuse soprattutto oltre i 2000 m s.l.m., consistono soprattutto di seslerieti, vegetazione di colonizzazione dei macereti dolomitici e pascoli a *Carex ferruginea*.

Le informazioni storiche indicano che la malga fu utilizzata fino al 1200 per il pascolamento ovino estensivo, nei secoli successivi per il pascolamento misto con ovini, bovini ed equini e intorno al 1980 la malga venne abbandonata.

Materiali e metodi

Analisi fitosociologica

Nel mese di luglio degli anni 1991-92, circa 25 ha di malga Padeon posti tra le quote 1815 e 1925 m s.l.m. sono stati rilevati con il metodo fitosociologico di Braun-Blanquet (1964) mediante 87 rilievi distribuiti su tutti gli aspetti vegetazionali presenti. Per ciascuna area di saggio è stata valutata la profondità del suolo quale media di cinque misure.

La matrice dei dati di presenza-assenza delle specie nei rilievi fitosociologici è stata sottoposta ad un procedimento di classificazione automatica per definire i tipi di vegetazione. Tale elaborazione è stata effettuata con il metodo della minima varianza (Feoli *et al.* 1982) utilizzando come funzione di somiglianza il *similarity ratio* (Westoff & van der Maarel 1978).

L'identificazione dei tipi vegetazionali è stata basata su dati di letteratura (Oberdorfer 1977, 1978, 1983b, 1992). La loro valutazione ecologica è stata ottenuta per via indiretta mediante gli indici

ecologici di Landolt (1977) e i valori di abbondanza-dominanza delle specie.

Allo scopo di studiare l'influenza delle variazioni micromorfologiche del suolo sulla vegetazione sono state analizzate le differenze floristiche tra rilievi adiacenti di dosso e di avvallamento.

Per individuare le correlazioni fra variazione floristica dei rilievi ed eventuali gradienti ecologici è stata eseguita un'analisi indiretta di gradiente basata sul metodo delle componenti principali applicato alla matrice delle distanze euclidee fra rilievi (Orlóci 1978, Feoli *et al.* 1982). La matrice delle distanze euclidee è stata ottenuta in base ai dati di presenza-assenza delle specie nei rilievi.

Analisi agronomica

Per l'analisi del valore agronomico dei tipi principali di pascolo sono stati effettuati rilievi vegetazionali con il metodo dell'analisi lineare (Daget & Poissonet 1971). Tale metodo viene normalmente adottato nella caratterizzazione agronomica di prati e pascoli per stimare l'abbondanza delle specie. Il campionamento con il metodo dell'analisi lineare è stato condotto rilevando in 100 punti equidistanti lungo una linea di 10 m la presenza delle specie; il valore di abbondanza di ogni specie (contributo specifico) è dato dalla sua percentuale di presenza nel rilievo lineare.

Allo scopo di individuare tipi agronomici di vegetazione (ecofacies), cioè gruppi di rilievi in cui sono predominanti le medesime specie vegetali e, quindi, relativamente omogenei per qualità agronomica (Jacquier & Jouglet 1976), i rilievi lineari sono stati classificati con il metodo del legame completo applicato alla matrice delle distanze euclidee fra i rilievi (Orlóci 1978, Feoli *et al.* 1982).

Per ogni rilievo lineare il valore agronomico è dato dalla media degli indici del valore foraggero delle singole specie (Stählin 1970, Klapp 1971) ponderata in base al loro contributo specifico. Per ogni ecofacies è stato calcolato il valore agronomico medio espresso secondo una scala da 0 a 100.

Inoltre, per tre delle ecofacies più importanti, alla fine delle stagioni vegetative 1991 e 1992 è stata stimata la produzione annua di sostanza secca tagliando e pesando l'erba presente su superfici di 18 mq.

Per analizzare l'influenza del tipo di campionamento impiegato, è stato eseguito un confronto fra i risultati ottenuti impiegando su 28 aree campione il metodo di rilevamento

fitosociologico e quello dell'analisi lineare. Tale confronto è basato sul calcolo del coefficiente di correlazione cofenetica tra i dendrogrammi di classificazione (metodo di classificazione: legame completo; funzione di somiglianza: distanza euclidea) dei due set di rilievi (Orlóci 1978) e sulla correlazione fra i valori agronomici delle 28 aree rilevate ottenuti dai dati quantitativi dei due tipi di campionamento.

La nomenclatura delle piante vascolari e delle briofite segue rispettivamente Pignatti (1981b) e Augier (1966). Unica eccezione è data da *Sesleria varia* (Jacq.) Wettst. per la quale viene usato il binomio *Sesleria albicans* Kit ex Schult.

Risultati e discussione

Definizione e valutazione fitosociologica dei tipi di vegetazione

Il dendrogramma ottenuto con il metodo di classificazione automatica è riportato in Fig. 3. Esso evidenzia otto gruppi principali di rilievi. Nella valutazione fitosociologica dei gruppi di rilievi il gruppo 4, che comprende più della metà dei rilievi eseguiti, è stato diviso in tre sottogruppi (4a, 4b, 4c di Fig. 3). La distribuzione sul territorio indagato dei tipi di vegetazione individuati è riportata nella mappa di Fig. 4.

Gruppo 1 (Tab. 1). Pascolo a *Poa alpina* (*Poion alpinae*).

Questo gruppo è costituito da rilievi eseguiti nella parte centrale dell'area dell'alpeggio su superfici quasi pianeggianti e con suolo profondo. Essi sono attribuibili all'alleanza *Poion alpinae* (Oberdorfer 1983b) per la presenza di molte specie caratteristiche di questo syntaxon (*Poa alpina*, *Crepis aurea*, *Trifolium badium*, *Trifolium pratense* subsp. *nivale* e *Phleum alpinum*). I rilievi in esame hanno una certa affinità floristica con l'associazione *Crepido-Festucetum rubrae* con cui, tuttavia, si preferisce non identificarli per la mancanza di *Plantago atrata*, *Ligusticum mutellina* e di *Alchemilla* sect. *Alpina*, specie tipiche dei pascoli pingui delle Alpi centro-settentrionali ma assenti in quelli delle Alpi meridionali. Per la presenza delle specie di alleanza citate i rilievi di Padeon sono simili a quelli effettuati da Lausi *et al.* (1981) nelle Alpi Giulie Occidentali e da Bezzi *et al.* (1984) nella catena del Brenta (Trentino Occidentale).

Il pascolo a *Poa alpina* di Padeon presenta elevati valori di copertura di *Deschampsia*

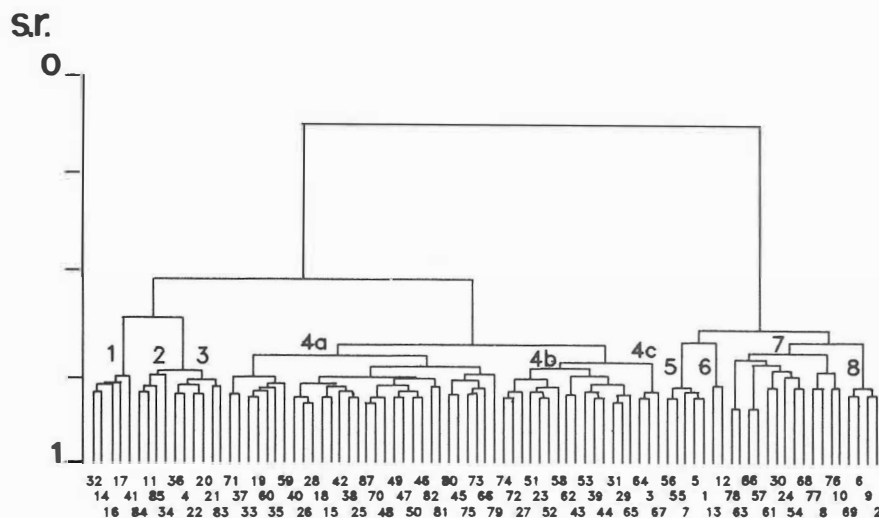
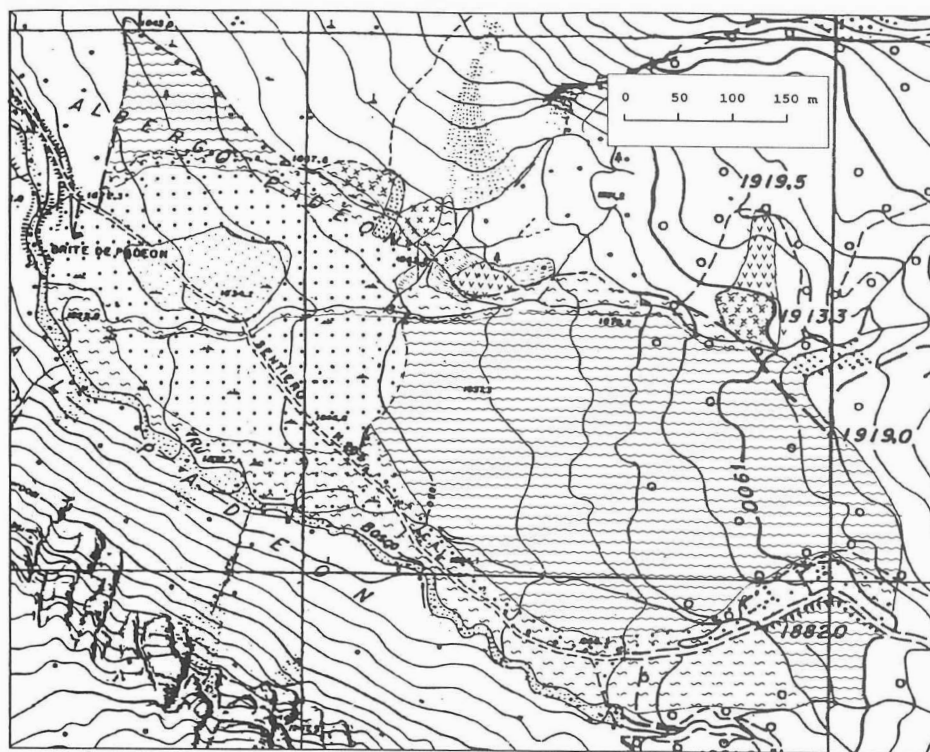


Fig. 3 - Dendrogramma dei rilievi fitosociologici; s.r.: similarity ratio.
Dendrogram of the phytosociological relevés; s.r.: similarity ratio.



- | | |
|--|---|
| 1 Pascolo a <i>Poa alpina</i> (<i>Poion alpinae</i>) 0.9 ha | 5+8 Mosaico tra pascolo a <i>Sesleria albica</i> . di bosco pascolato e bosco a larice e cembro (<i>Vaccinio-Pinetum cembrae rhododendretum hirsuti</i>) 12.2 ha |
| 2 Torbiera a <i>Carex davalliana</i> (<i>Caricetum davallianae</i>), forma tipica 0.2 ha | 6 Vegetazione dei ghiaioni dolomitici (<i>Thlaspietea rotundifolii</i>) 0.1 ha |
| 3 Torbiera a <i>Carex davalliana</i> (<i>Caricetum davallianae</i>), in via di prosciugamento 0.3 ha | 7 Boscaaglia a mugo (<i>Erico-Rhododendretum hirsuti</i>) 4.5 ha |
| 4a e 4b Pascolo a <i>Carex ferruginea</i> (<i>Caricetum ferrugineae trifolietosum</i>) 4.5 ha | Mosaico tra boscaaglia a mugo (<i>Erico-Rhododendretum hirsuti</i>) e pascolo a <i>Carex ferruginea</i> (<i>Caricetum ferrugineae trifolietosum</i>) 1.1 ha |
| 4c Pascolo a <i>Sesleria albicans</i> (<i>Carici ornithopodae-Seslerietum albicans</i>), forma tipica 0.3 ha | Mosaico tra boscaaglia a mugo (<i>Erico-Rhododendretum hirsuti</i>) e pascolo a <i>Sesleria albicans</i> (<i>Carici ornithopodae-Seslerietum albicans</i>) 0.7 ha |

Fig. 4 - Mappa vegetazionale di Malga Padon.
Vegetation map of Malga Padon.

Tab. 1 - Pascolo a *Poa alpina* (gruppo di rilievi 1 di Fig. 3).
Altitudine: 1830-1840 m; superficie ril.: 100 mq; copertura
vegetale: 100 % (75 % nel ril. 16).

Poa alpina pasture (relevé group 1 of Fig. 3).

Gruppo rilievi	1					PRE- SENZE
Numero rilievo	3	1	1	1	4	
	2	4	6	7	1	
Esposizione	S	S	S	S	O	
Inclinazione (%)	0	0	0	0	0	
	1	1	5	5	5	
N. totale specie	2	2				
	3	4	5	3	2	
Profondità del suolo (cm)	4	6	2	4	6	
	1	1	1	2	2	
	5	6	6	1	3	
POION ALPINA						
Phleum alpinum	1	+	1	1	1	5
Trifolium pratense/nivale	1	1	2	+	+	5
Poa alpina	1	+	+	1	+	4
Crepis aurea		+	+	+	+	2
Trifolium badium		+	+			2
Ranunculus montanus			+	+		2
ARRHENATHERETEA						
Alchemilla vulgaris	2	+	1	2	1	5
Carum carvi	1	+	+	1	1	5
Achillea millefolium	1	1	1	1	1	5
Avenula pubescens	+	2	1	1	1	5
Dactylis glomerata	1	2	2	1	1	5
Veronica chamaedrys	1	1	1	2	1	5
Leontodon hispidus	+	+	+	+		4
Heracleum sphondylium	+	1	1	1		4
Trifolium repens	+	+	+		+	4
Crocus albiflorus		+	+	+	+	3
Anthriscus sylvestris		+	+	+		2
Trisetum flavescens	1	2				2
Vicia sepium			1		1	2
MOLINIO-ARRHENATHERETEA						
Festuca pratensis	2	2	3	1	1	5
Rhinanthus freynii	1	1	2	+	1	5
Taraxacum alpinum	2	1	1	1	+	5
Ranunculus acris	2	1	1	1	1	5
Rumex acetosa	1	1	1	1	1	5
Festuca rubra	1		+	+	+	4
Leucanthemum adustum	+	+	+			3
Poa pratensis		1	+		1	3
Galium mollugo			1		1	2
Poa trivialis		1	+			2
Trollius europaeus				+	1	2
SPECIE COMPAGNE						
Campanula scheuchzeri	+	+	+	1	1	5
Deschampsia caespitosa	3	2	2	4	4	5
Urtica dioica	+	2	1	1	1	5
Senecio gaudinii	2	2	2		1	4
Myosotis alpestris	+	+	+	+		4
Carex sempervirens	+		1	+		3
Carduus carlinaefolius	+	+	1			3
Phyteuma orbiculare	+		+	+		3
Plantago media		+	+	+		3
Briza media		2	1	1		3
Gentianella engadinensis	+	+	+			3
Chenopodium bonus-henricus	1	1	1			3
Potentilla crantzii		+	+			2
Carlina acaulis		+	1			2
Polygonum viviparum			+	+		2
Silene vulgaris	1		1			2
Knautia longifolia			+	+		2
Petasites albus			1	1		2
Galium pumilum	+	1				2

caespitosa. La massiccia presenza di tale graminacea in questo tipo di vegetazione sembra dovuta soprattutto al tipo di terreno, che presenta tessitura tendenzialmente argillosa, elevata capacità idrica e buon contenuto di elementi nutritivi, fattori questi corrispondenti alle note esigenze ecologiche della

specie (Rieder *et al.* 1976). La diffusione di tale specie nell'area indagata potrebbe essere dovuta anche all'abbandono del pascolamento data la sua elevata capacità concorrenziale.

Gruppo 2 (Tab. 2). Formazione a *Carex davalliana* (*Caricetum davallianae*) su suoli poco drenati.

I rilievi di questo gruppo sono localizzati in aree del pascolo con falda acquifera superficiale. Essi sono attribuibili all'associazione *Caricetum davallianae* (Oberdorfer 1977) per la presenza di numerose specie caratteristiche quali: *Triglochin palustris*, *Carex fusca*, *Campylopus stellatum*, *Parnassia palustris*, *Juncus alpino-articulatus*, *Primula farinosa* e *Carex lepidocarpa*, *Tofieldia calyculata* e *Carex davalliana*. Rispetto alle due forme dell'associazione citate da Oberdorfer (1977), e cioè quella montana e quella subalpino-alpina, i rilievi in questione appaiono floristicamente più simili alla seconda per la presenza di *Equisetum variegatum*, *Selaginella selaginoides* e *Willemetia stipitata*. Accanto alle specie succitate sono presenti numerose specie caratteristiche di *Molinio-Arrhenatheretea* e *Seslerietea albicantis*. A nostro avviso tale presenza è dovuta alle particolari caratteristiche morfologiche delle aree rilevate: pendio poco scosceso costituito da un'alternanza di leggere depressioni e di piccole prominenze. Tale micromorfologia del terreno determina un mosaico vegetazionale costituito da formazioni igrofile (*Caricetum davallianae*) negli avvallamenti e da formazioni meno igrofile sui dossi, sui quali sono presenti pure molte specie di *Scheuchzerio-Caricetea fuscae*.

Il rilievo 34, localizzato in prossimità di un piccolo corso d'acqua e dominato da *Cratoneurum commutatum* subsp. *falcatum* è attribuibile all'associazione *Cratoneuretum falcati*, tipica degli orizzonti subalpino e alpino su substrato calcareo in vicinanza di risorgive (Oberdorfer 1977).

Gruppo 3 (Tab. 2). Formazione a *Carex davalliana* (*Caricetum davallianae*) su suoli maggiormente drenati.

I rilievi di questo gruppo sono localizzati su terreni in lieve pendenza ed in prossimità delle aree più umide del gruppo precedente. Per la copresenza di diverse specie di *Scheuchzerio-Caricetea fuscae* e di numerose specie di *Molinio-Arrhenatheretea* e *Seslerietea albicantis* questa formazione può venir considerata una forma di transizione tra il *Caricetum davallianae* (Gruppo 2) e i tipi di pascolo seguenti.

Tab. 2 - Formazione a *Carex davalliana*, su suoli poco drenati (gruppo di ril. 2 di Fig. 3) e su suoli maggiormente drenati (gruppo di ril. 3 di Fig. 3). Altitudine: 1850-1920 m; superficie ril.: 100 mq (75 mq nel ril. 85); copertura vegetale: 100 % (50 % nel ril. 20).

Carex davalliana community, on poorly drained soil (relevé group 2 of Fig. 3) and on highly drained soil (relevé group 3 of Fig. 3).

Gruppo di rilievi	2				3					
Numero rilievo	84	11	85	34	36	4	22	20	21	83
Esposizione	SO	SO	SE	SO	SO	SO	SO	SO	SO	SO
Inclinazione (%)	17	10	10	7	10	8	20	10	23	9
N. totale specie	39	36	38	40	55	65	61	60	59	40
Profondità del suolo (cm)	63	67	60	12	34	44	22	43	41	66

CARICETUM DAVALLIANAE e CARICION DAVALLIANAE										
<i>Carex davalliana</i>	2	2	2	2	2	2	+	2		IV
<i>Tofieldia calyculata</i>	1	+								I
<i>Eriophorum latifolium</i>								+		I
TOFIELDIETALIA e SCHEUCH- ZERIO-CARICETEA FUSCAE										
<i>Carex lepidocarpa</i>	2	1	2	+	+	+	+	2	2	V
<i>Carex panicea</i>	2	2	2	2	2	1		2	1	IV
<i>Primula farinosa</i>	+	+	+	+	+	+	+	+	+	IV
<i>Wilmetia stipitata</i>	2	2	2		2	2				III
<i>Parnassia palustris</i>		+						+	+	III
<i>Carex fusca</i>	2		2			1	2		1	III
<i>Juncus alpino-articulatus</i>			+				2			II
<i>Selaginella selaginoides</i>	+	+								I
<i>Triglochin palustre</i>			+							I
<i>Campyllum stellatum</i>							1			I
SPECIE COMPAGNE										
<i>Trifolium repens</i>	+	+	+	+	+	+	+	+	+	V
<i>Ranunculus acris</i>	1	1	+	1	+	2	2	2	1	V
<i>Leontodon hispidus</i>	2	1	2	+	2	1	2	+	2	V
<i>Alchemilla vulgaris</i>	+	+	+	2	2	2	+	+	1	V
<i>Crepis aurea</i>	1	1	1	+	1	1	1	1	+	V
<i>Rhinanthus freynii</i>	2	1	+	+	+	+	+	1	2	V
<i>Carex ferruginea</i>	2	2	1	+	2	2	1	+	2	V
<i>Potentilla erecta</i>	1	1	+	+	+	+	+	2	1	V
<i>Briza media</i>	2	1	2	+	2	1	2	1	2	V
<i>Deschampsia caespitosa</i>	1	1	+	2	+	1	2	2	2	V
<i>Polygonum viviparum</i>	1	+	+	1	1	+		+	1	V
<i>Trifolium pratense/nivale</i>	1	+	+		1	+	1	+	2	V
<i>Carex sempervirens</i>	2	2	2		1	1	+	+	2	V
<i>Poa alpina</i>	1		1		+	+	1	+	+	IV
<i>Festuca rubra</i>	1		1		1	1	1	2	1	IV
<i>Lotus alpinus</i>	+	1			+	1	2	1	1	IV
<i>Polygala alpestris</i>	+	+			+	+	+	+	+	IV
<i>Prunella grandiflora</i>	1		2		1	2	+	+	2	IV
<i>Soldanella alpina</i>	+	+			+	+	+	+	+	IV
<i>Carex flacca</i>		2			+	2	2	1	+	IV
<i>Trollius europaeus</i>	1	1	+		+					III
<i>Nardus stricta</i>	2	2	2		2	2				III
<i>Equisetum variegatum</i>	1	2		+	+	+				III
<i>Ranunculus montanus</i>	1	+			+	+	+	+		III
<i>Sesleria albicans</i>	1		1		+	2		+	2	III
<i>Phyteuma orbiculare</i>		+	+		+	+			1	III
<i>Carex pallens</i>	2		1		+	1		+	1	III
<i>Aposperis foetida</i>	1		+		+	+	1	1		III
<i>Carum carvi</i>	+				+	1	+	+	2	III
<i>Trifolium badii</i>			+		+	1	+	1	1	III
<i>Leucanthemum adustum</i>	2				+	+	+	+	1	III
<i>Lathyrus pratensis</i>		1			+	2	+	+		III
<i>Campanula scheuchzeri</i>		+			+	+	+	+	+	III
<i>Plantago media</i>		+			+	+	+	+	+	III
<i>Petasites albus</i>			1		2	+	2	+	+	III
<i>Luzula multiflora</i>		+			+	+	+	+	1	III
<i>Anthoxanthum alpinum</i>		2			1	+	1	1	1	III
<i>Carex ericetorum</i>		+			+	1	+	+		III
<i>Linum catharticum</i>					+	+	1	+	+	III
<i>Taraxacum alpinum</i>					+	+	+	+	2	III
<i>Anthrillia vuln./alpestris</i>					1	+	+	2	1	III
<i>Galium pumilum</i>					+	+	+	+	+	III
<i>Agrostis tenuis</i>					+	+	1	+	2	III
<i>Carex tomentosa</i>	2	2	2							II
<i>Cerastium holosteoides</i>		+	+		+			+		II
<i>Caltha palustris</i>	1		1		1			1		II
<i>Molinia caerulea</i>	2	1			+					II
<i>Gymnadenia conopsea</i>	1				1	+			+	II
<i>Gentiana verna</i>	+				+	1	+			II
<i>Carex capillaris</i>		1			+	+	+	+		II
<i>Scabiosa lucida</i>			+			1	2			II
<i>Hippocrepis comosa</i>	2				+	+				II
<i>Thesium alpinum</i>		+			+	+		+	+	II
<i>Gentianella engadinensis</i>		+			+	+				II
<i>Melampyrum sylvaticum</i>		+			+			2		II
<i>Vaccinium vitis-idaea</i>	1				+			1		II
<i>Veronica chamaedrys</i>						1	+	+		II
<i>Pedicularis verticillata</i>					+	+	+	+	1	II
<i>Gentiana ciliata</i>					+	+	+	+		II
<i>Coeloglossum viride</i>					+	+	+	+		II

Gruppi 4a e 4b (Tab. 3). Pascolo a *Carex ferruginea* (*Caricetum ferrugineae trifolietosum*).

Questo tipo di pascolo è il più diffuso nell'area indagata. I rilievi sono localizzati su lievi pendii con morfologia più o meno ondulata per la presenza di dossi e di depressioni profonde fino a 40 cm.

Per l'elevata frequenza di *Carex ferruginea* e per la presenza di *Pimpinella major* e *Trollius europaeus*, specie differenziali di *Caricion ferrugineae*, questo tipo di pascolo è stato attribuito all'associazione *Caricetum ferrugineae*. Delle due subassociazioni descritte da Oberdorfer (1978), i rilievi di Padeon, presentando costantemente *Trifolium repens*, *Alchemilla vulgaris* e *Plantago media*, potrebbero venir considerati come appartenenti alla subassociazione *trifolietosum* rispetto alla quale, tuttavia, sono caratterizzati da una maggiore ricchezza floristica dovuta alla presenza di più numerose specie di *Seslerietea albicantis* e di *Molinio-Arrhenatheretea*.

Il grado con cui le specie caratteristiche delle classi fitosociologiche succitate sono rappresentate nei rilievi è variabile. Infatti, i rilievi del gruppo 4b, spesso localizzati sul fianco o al margine di doline, presentano mediamente un minor numero di specie di *Molinio-Arrhenatheretea* e un maggior numero di specie di *Seslerietea albicantis* rispetto a quelli del gruppo 4a.

Gruppo 4c (Tab. 4). Pascolo a *Sesleria albicans* (*Carici ornithopodae-Seslerietum albicantis*).

Questo tipo di vegetazione si sviluppa su pendii con suolo poco profondo. Nei rilievi predominano specie di *Seslerietea albicantis*; sono inoltre presenti numerose specie di *Molinio-Arrhenatheretea* e di *Festuco-Brometea*. In base alla recente analisi delle praterie naturali del Friuli su substrato calcareo-dolomitico (Chiappella Feoli & Poldini 1993), tali rilievi possono venir attribuiti all'associazione *Carici ornithopodae-Seslerietum albicantis*, suballeanza *Ranunculenion hybridi*, alleanza *Caricion austroalpinae*. Quest'ultima unità è rappresentata nel nostro caso da due specie caratteristiche (*Laserpitium peucedanoides*, *Horminum pyrenaicum*) sulle dieci citate dagli autori, mentre è presente una delle nove specie caratteristiche della suballeanza *Ranunculenion hybridi*, e cioè *Pedicularis elongata*. In base alle considerazioni svolte dagli autori citati, la scarsità delle specie di alleanza e di suballeanza sarebbe da attribuire al fatto che i rilievi qui riportati provengono da un'area decentrata verso nord rispetto a quella di ottimale sviluppo dei *syntaxa* menzionati e in cui gli stessi

- Pascoli subalpini di Malga Padeon -

Tab. 3 - Pascolo a *Carex ferruginea*; sono riportati in tabella solo 17 dei 45 rilievi appartenenti ai gruppi di rilievi 4a e 4b di Fig. 3.
 Altitudine: 1820 - 1850 m; superficie ril.: 100 mq (40 mq nel ril. 28, 60 mq nei ril. 27 e 39); copertura vegetale: 100 % (85 % nel ril. 25).
Carex ferruginea pasture; only 17 of the 45 relevés belonging to relevé groups 4a and 4b of Fig. 3 are reported.

Gruppo di rilievi	4a												4b					CLASSE
Numero rilievo	37	33	28	15	25	70	47	50	82	80	66	79	27	52	43	39	31	DI PRE-
Esposizione	SO	SO	SO	SO	SO	NO	NO	NO	SO	O	SO	O	SO	NO	NO	SO	SO	SENZA
Inclinazione (°)	5	10	10	10	10	8	5	10	5	5	15	5	10	4	15	10	10	
N. totale specie	38	55	53	62	62	52	59	50	40	38	53	21	36	60	43	46	63	
Profondità del suolo (cm)	15	14	15	14	7	14	17	19	14	15	13	16	8	16	9	6	13	
CARICETUM FERRUGINEAE																		
e CARICION FERRUGINEAE																		
Trollius europaeus	1		1	+	+	+	1	1	1	2	+		+		+	+		IV
Carex ferruginea	1	1		2	+	1	1	+			1		1	+	1	+		IV
Pimpinella major		+																I
SESLERIETALIA CAERULEAE																		
e SESLERIETEA ALBICANTIS																		
Carex sempervirens	2	2	2	2	2	2	2	2	2	2	1	+	2	2	2	2	2	V
Horminum pyrenaicum	2	2	1	2	2	1	1	1	2	1	2	1	2	2	2	2	2	V
Lotus alpinus	1	1	+	1	1	+	1	+	1	1	1		+	1	1	1	+	V
Sesleria albicans	2	2	+	1	2	1	2	1	2	2	2		2	2	2	2	1	V
Carduus carlinaefolius		1	+	1	1	1	1	1	2	2	1	1	2		1	1	1	V
Scabiosa lucida		1	1	+	1	+	1	2			1		2	2	+	1	1	V
Phyteuma orbiculare		1	+	+	+	1	+	+	1	+			+	+	+	+		IV
Campanula scheuchzeri		+	+	+	+	+	+	+	+	+			+	+	+	+		IV
Anthyllis vulneraria/alpestris		+	+	1	2		1		1	2			2	2	2	2	1	IV
Pedicularis elongata		1	+	+	+	+			+				1	+	2	1	1	IV
Gentiana verna		+	+	+	+	1			1	+			1	+	+	1	+	IV
Potentilla crantzii		+	+	1	+		+	+	+				1	+	+	1	+	IV
Polygala alpestris		+	+	+	+	+	+	+	+	+			+	+	+	1	+	IV
Biscutella laevigata		+		2		2			1	+	1		+	+	+		1	III
Aster bellidiastrium					+		1	1					+			1	+	II
Erigeron polymorphus					+	1							+	+		1	+	II
Helianthemum oelandicum/alpestre					+								1	+	+	1	+	II
Carex atrata					1		1	1	1	1	+							II
Acinos alpinus					+						+		+					I
Pedicularis verticillata						+		+					+					I
Cerastium alpinum/alpinum				+	+													I
Stachys alopecurus											+					1	+	I
Hieracium villosum																	+	I
Arabis ciliata																	+	I
Dryas octopetala															+			I
SPECIE COMPAGNE																		
Prunella grandiflora	2	2	+	1	2	1	1	+	2	2	2	1	2	1	2	2	1	V
Polygonum viviparum	1	1	+	2	+	2	+	1	1	1	1	+	+	+	+	+	1	V
Trifolium pratense/nivale	1	+	+	+	1	1	1	1	1	1	+		1	1	+	+	+	V
Leucanthemum adustum		+	+	+	+	1	+	+	1	1	1	1	+	+	+	2	+	V
Briza media	1	2	+	1	+	1	2	2	1	1	2		2	2	2	2	2	V
Rhinanthus freynii	2	+	1	2	1	1	1	2	1	1	1	1	1	+	1	1	+	V
Ranunculus acris	1	+	1	1	1	1	1	1	1	+	+	1	+	+		1		V
Galium pumilum	1	+	+	1	1	1	+	+	1	+	+		+	+	+	1	+	V
Soldanella alpina		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	1	V
Avenula pubescens		+	1	1	1	1	+	1	1	1	1	1	+	+	+	1	+	V
Plantago media	1	1	+	1	+	+	+	+	1	1			1	+		1	+	V
Crocus albiflorus	1		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	IV
Agrostis tenuis			2	1	1	1	2	2	1	2	1	2	+	+	+	2		IV
Anthoxanthum alpinum			1	1	1	2	1	1	1	+	1	1		2		1	1	IV
Poa alpina		1	+	1	1	1	1	1	1	1	1		1	+		+		IV
Trifolium repens	1	+	+	+	+	+	+	+	+	+	+		+	+				IV
Alchemilla vulgaris	1	+	1	1	1	+	1	2		1		2		+			1	IV
Carum carvi	1	1	1	1	+	+	+	1	1				+		+	1		IV
Achillea millefolium	1	+	1	1	+	1	1	1	1	1	2						1	IV
Carlina acaulis	1	1	1	1	+	1	1		1	1	+		1		2	1	1	IV
Thymus alpinus		+	+	+	2	1	+	+	1	+	+		+	+		1	+	IV
Luzula multiflora				+	+	+	+	+	1	+	+		+	+		+	+	IV
Cirsium acaule	1	1	2	+	+	1	1		2	+			1			1	1	IV
Ranunculus montanus	1	+	+	+	+	+	+	+	+							+	1	IV
Deschampsia caespitosa	2	1	1	1	+	1	2	2		1		4		1		+	+	IV
Selaginella selaginoides		+			+			+					+	+	+	+	+	III
Festuca rubra		1	1	2		+	1	1		1	+			1		2		III
Carex pallescens			1	+			1	1	+	+	+		+	+				III
Knautia longifolia		+	2	+			+		1		+						1	III
Myosotis alpestris				+	+	+	+	1										III
Silene vulgaris	1	+	+	+	+	+	+	1				2					1	III
Carex flacca	2	1				+	1	2			+	+						III
Leontodon hispidus	1	2					1	+	1					2	+			III
Koeleria pyramidata		+			1								2	1	1	1	1	III
Primula farinosa			+						+					+	+	1	+	III
Aposperis foetida									+		1							III
Thesium alpinum		+	+															III
Veronica chamaedrys			+	+			+	+				1						III
Festuca valesiaca	1				2	1						2			2		+	III
Potentilla erecta						+					+			+	+	+	+	III
Hippocrepis comosa	1			+	1													II
Gentianella engadinensis	1	+			+				+									II
Cerastium holosteoides						1	+	+	+	+	+							II
Tortella tortuosa						+	+	+								+	+	II
Carex ornithopoda						1	+	+									+	II
Festuca pratensis	1	+	1														1	II
Rumex acetosa																		II
Dactylis glomerata	1	1	1						1									II
Phleum alpinum		+	1	+					1		+							II
Homogone alpina																1	+	II
Gymnadenia conopsea														+	+			II
Crepis aurea	1								+	+						1		II
Petasites albus	1	2																II
Taraxacum alpinum	+	+																II
Senecio gaudinii					1	2												II
Geum rivale					+			+	+	+								II

Tab. 4 - Pascolo a *Sesleria albicans* (gruppo di ril. 4c di Fig. 3), lo stesso in bosco pascolato (gruppo di ril. 5 di Fig. 3), e vegetazione al margine di ghiaioni dolomitici (gruppo di ril. 6 di Fig. 3). Altitudine: 1860-1940 m; copertura vegetale: 100 % (90% nel ril. 67, 25% nel ril. 12, e 30% nel ril. 13).

Sesleria albicans pasture in open areas (relevé group 4c of Fig. 3) and in grazed woodland (relevé group 5 of Fig. 3), and vegetation on the margins of dolomitic screes (relevé group 6 of Fig. 3).

Gruppo di rilievi	4c			5					CLASSE	6		PRE-
Numero rilievo	64	3	67	7	5	1	56	55	DI PRE-	12	13	SENZA
Esposizione	SO	SO	SO	SO	SO	SO	SO	NO	SENZA	SO	SO	
Inclinazione (%)	10	12	7	11	14	12	10	4		10	10	
N. totale specie	62	64	67	54	51	43	51	56		28	28	
Profondità del suolo (cm)	5	12	3	25	17	9	9	11		1	1	
CARICI ORNITHOPODAE-SESLERIETUM												
ALBICANTIS E RANUNCULENTION HYBRIDI												
Carex ornithopoda	+	+	+	1	+	1	+	+	V			
Pedicularis elongata	+	+	+	1	1	1	+	+	V			
CARICION AUSTRALPINAE												
Horminum pyrenaicum	1	2	+	2	+	1	2	2	V			
Laserpitium peucedanoides		1	+	+	+	1	1	+	V			
SESLERIETALIA CAERULEAE												
SESLERIETEA ALBICANTIS												
Sesleria albicans	1	2	2	2	2	1	2	2	V	+	+	2
Phyteuma orbiculare	+	+	+	+	+	+	+	1	V			
Biscutella laevigata	+	1	+	+	+	1	+	1	V			
Carex sempervirens	2	1	1	1	1	2	2	2	V			
Lotus alpinus	1	1	1	+	+	+	+	1	V	1	2	2
Carduus carlinaefolius	1	2	1	1	1	1	+	1	V	1	1	2
Campanula scheuchzeri	+	+	+	+	1	+	+	+	V	1	1	2
Anthyllis vuln./alpestris	2	+	1	+	+	+	1	2	V	1		1
Polygala alpestris	+	+	+	+	+	+	+	+	IV	1	+	2
Gentiana verna	+	+	1	+	+	+	+	+	IV			
Helianthemum oeland./alpestre	+	+	1	+	+	+	1	+	IV			
Aster bellidiastrium			+	+	1	+	+	+	IV			
Carex ferruginea				3	1	2	2	2	IV			
Pedicularis verticillata	+	+	1					+	III			
Scabiosa lucida	+	+		+				+	III			
Hieracium bifidum			+	1	1	1			III	+	+	2
Erigeron polymorphus	1	+	+						II	+		1
Dryas octopetala			+				+	+	II	+		1
Potentilla crantzii				+		1			II		+	1
SPECIE COMPAGNE												
Ranunculus acris	+	+	1	+	+	1	+	1	V			
Galium pumilum	+	+	+	+	+	1	+	1	V	1	2	2
Polygonum viviparum	1	+	+	+	1	1	+	+	V		+	1
Poa alpina L.	+	+	+	+	+	+	+	1	V	1	+	2
Carlina acaulis	2	1	2	+	+	+	1	+	V	+	1	2
Silene vulgaris	1	1	2	+	1		2	+	V	4	3	2
Thymus alpinus	1	+	+	+	+	+	+	+	V	2	2	2
Erica carnea		+	+	+	2	2	1	1	V	+		1
Festuca rubra	1		2	+	+	1	2	1	V			
Potentilla erecta	+	+		1	1	1	1	+	V			
Aposeris foetida	1	1		1	3	2	+	1	V			
Prunella grandiflora	1	1	+	+			2	+	IV			
Euphrasia pulchella	1	+	+	+			1	1	IV			
Selaginella selaginoides		+	+	+	+	+	+	+	IV			
Homogone alpina				+	1	1	+	2	IV			
Melampyrum sylvaticum			+	+	1	1	1	2	IV			
Daphne striata			+	+	+	1	2	1	IV			
Soldanella alpina		+		+	1	1	+	1	IV			
Leontodon hispidus	1	2	+	1				1	IV	1	+	2
Parnassia palustris	1	+	+	+		1			IV			
Ranunculus montanus	+	+		+	1	1	+		IV	+		1
Juniperus communis/montana			+	1	+	1	+		IV			
Trollius europaeus			+	1	+	+	1	1	IV			
Tofieldia calyculata			1	+	+		1	1	IV			
Vaccinium vitis-idaea				1	2	2	+	1	IV			
Trifolium repens	+	+	+					+	III		+	1
Leucanthemum adustum	1	1	1					2	III	+		1
Trifolium pratense/nivale	1	1	1					+	III			
Rhinanthus freynii	1	1	1	+					III			
Hippocrepis comosa	2	1	1	1					III			
Gentianella engadinensis		+	+					+	III	+		1
Briza media	1	2	2					1	III			
Thesium alpinum	r	r	r				r		III			
Carum carvi	+		+		1			+	III	+	+	2
Crepis aurea		+	+					+	III			
Solidago virga-aurea			r	r	r	r			III			
Vaccinium myrtillus				+	1	2		1	III			
Luzula sieberi/sieberi				2	2	2		1	III			
Anthoxanthum alpinum				1	+		+	1	III			
Linum catharticum	1	+	+						II		+	1
Plantago media		+	+						II		+	1
Gentiana ciliata	1	+	+						II	1		1
Orobancha gracilis	1	r	r						II			
Hieracium piloselloides	1	+	+						II			
Pinus cembra		1					1	1	II			
Calamagrostis villosa				1	2	2			II			
Cirsium erisithales				1	1	+			II			
Galium mollugo				+	+	+			II			
Picea excelsa		1	1						II	+		1
Hieracium staticifolium	1		+						II	2	2	2
Acinos alpinus	1	1							II		1	1
Petasites albus			2				2		II		+	1
Tortella tortuosa		+				1			II	+		1
Festuca valesiaca		2							I		+	1
Arabis pumila										+	+	2
Minuartia verna										+	2	2
Hutchinsia alpina										+		1
Linaria alpina										+		1
Veronica aphylla										+		1
Achillea millefolium											1	1
Trifolium badium											1	1
Cerastium holosteoides											1	1

lascerebbero il posto all'alleanza *Seslerion caerulae*. Dell'associazione di riferimento, infine, i rilievi in esame possiedono la specie differenziale *Carex ornithopoda* mentre mancano della specie caratteristica *Ranunculus venetus*. Secondo i succitati autori l'associazione in questione costituirebbe una cenosi di transizione tra *Seslerietea albicantis* e *Festuco-Brometea* e sarebbe caratterizzata da numerose specie di quest'ultima classe. Tale presenza caratterizza pure i rilievi di Padeon anche se in misura minore rispetto a quelli riportati dagli autori citati e ciò probabilmente perchè la loro quota è di almeno 250 m superiore.

Gruppo 5 (Tab. 4). Pascolo a *Sesleria albicans* (*Carici ornithopodae-Seslerietum albicantis*) in bosco.

I rilievi di questo gruppo sono rappresentativi delle aree a vegetazione prevalentemente erbacea che nel bosco pascolato si alternano a mosaico con le aree dominate dalla vegetazione arbustivo-arborea. Nei rilievi in esame prevalgono le specie di

Seslerietea albicantis, ma ben rappresentata è anche la classe *Vaccinio-Piceetea*. Rispetto al pascolo a *Sesleria albicans* (Gruppo 4c) si differenzia per la cospicua presenza di specie nemorali delle classi *Vaccinio-Piceetea* e, in minor misura, *Erico-Pinetea* e per la ridotta incidenza di specie di *Molinio-Arrhenatheretea*. Per tali caratteristiche è possibile interpretare il gruppo di rilievi in esame come uno stadio di transizione tra il *Carici ornithopodae-Seslerietum albicantis* e il bosco a pino cembro con cui tali rilievi sono in continuità topografica. Tale transizione sembra favorita dall'interruzione del pascolamento e, in base alle osservazioni effettuate, avviene mediante la progressiva sostituzione delle specie del pascolo a *Sesleria albicans* soprattutto con *Vaccinium vitis-idaea*, *V. myrtillus* ed *Erica carnea*.

Gruppo 6 (Tab. 4). Gruppo 6 (Tab. 4). Vegetazione al margine di ghiaioni dolomitici (*Thlaspietea rotundifolii*).

Questo gruppo è costituito da rilievi localizzati ai margini di un conoide di deiezione, che con i suoi

Tab. 5 - Mugheta (gruppo di ril. 7 di Fig. 3). Altitudine: 1840-1900 m; copertura vegetazione: 100 %.
Pinus mugo community (relevé group 7 of Fig. 3).

Gruppo rilievi	7														CLASSE DI FRE- QUENZA
Numero rilievo	78	63	86	57	61	30	24	54	68	77	8	76	10		
Esposizione	SO	SO'	SO	SO	SO	SO	SO	NO	SO	SO	SO	SO	SO		
Inclinazione (%)	15	15	10	10	5	10	5	5	10	10	10	15	15		
Superficie di rilievo (mq)	100	40	100	125	40	15	30	30	100	90	100	100			
N. totale specie	14	17	18	12	17	12	18	13	17	14	28	34	23		
Profondità del suolo (cm)	12	11	17	13	14	10	16	17	2	11	21	7	13		
ERICO-RHODODENDRETUM HIRSUTI															
Pinus mugo	4	3			4	3	4	4	5	4	4	1	3	2	
Homogone alpina	1	1	+		1	1					1	1	1		
Daphne striata			2	1			2	+			1	2			
Vaccinium vitis-idaea			+			1					2	2	1		
Vaccinium myrtillus											2	2			
ERICO-PINETALIA															
ED ERICO-PINETEA															
Erica carnea				3	2			+			2	2		1	
Cirsium erisithales									2		r	1	+	+	
SPECIE COMPAGNE															
Sesleria albicans	2	2	+			2	2	1	1	2	1	1	2	1	
Aposotis foetida	1	1	1	2		2	2		1		1	1	1	+	
Melampyrum sylvaticum	+	+	+	1	1	1	2				+	1	+		
Trollius europaeus	1	1	+	1	2	1	1	2							
Potentilla erecta	1	1	+	1		+		1					1	1	
Pinus cembra			2	2		2		1				2	2	2	
Picea excelsa			2	2	2				2			2	2	4	
Carex sempervirens								1	1		1	1	1	+	
Aster bellidiastrium	+	+						1	1		1	1			
Anthyllis vulneraria/alpestris	+	+						1	+	1					
Stachys alopecurus	1	2				+							1	+	
Laserpitium peucedanoides			1					1			1	1	1		
Carex ferruginea								1			1	1			
Pedicularis elongata						1		1			1	1			
Carduus carlinaefolius						1	1	+			+	1			
Soldanella alpina						1					+	+	+		
Polygonum viviparum						1		+							
Thymus alpinus			+						1			1	+		
Larix decidua				1				1							
Juniperus communis var. montana				3				2				3			
Helianthemum oelandicum/alpestre	1	1													
Hieracium villosum	1	1										+			
Pedicularis verticillata						1	+	+					1		
Festuca rubra											+	1	1		
Ranunculus acris								2				1			
Prunella grandiflora						1						1	1		
Selaginella selaginoides									+			1	+		

lembi inferiori interessa la parte alta del pascolo. La loro copertura erbacea discontinua è formata da un numero ridotto di specie appartenenti soprattutto alla classe *Seslerietea albicantis*. Le specie della classe *Thlaspietea rotundifolii* sono *Acinos alpinus*, *Hutchinsia alpina*, *Hieracium staticifolium* e *Linaria alpina*. In misura ridotta risultano presenti anche specie delle classi *Molinio-Arrhenatheretea* e *Festuco-Brometea*.

Gruppo 7 (Tab. 5). Mugheta (*Erico-Rhododendretum hirsuti*).

La composizione floristica dei rilievi di questo gruppo è caratterizzata dalla presenza di diverse specie delle classi *Erico-Pinetea* e *Vaccinio-Piceetea* e dalla diminuzione delle specie di *Seslerietea albicantis* e dalla quasi totale scomparsa delle entità di *Molinio-Arrhenatheretea*. Questi rilievi possono venir ascritti all'associazione *Erico-Rhododendretum hirsuti*. Tale associazione è caratterizzata dalla dominanza di *Pinus mugo* e di *Rhododendron hirsutum* e dalla presenza di un ampio corteggio di specie basifile (Oberdorfer 1992). Le specie caratteristiche e differenziali di questa associazione presenti anche nei rilievi di Padeon sono *Daphne striata*, *Pinus mugo*, *Vaccinium vitis-idaea*, *V. myrtillus* e *Homogine alpina*, mentre è sempre assente *Rhododendron hirsutum*.

A differenza della caratterizzazione dell'*Erico-Rhododendretum hirsuti* fatta da Oberdorfer (1992) secondo il quale nell'orizzonte subalpino-alpino delle Alpi calcaree tale associazione costituisce la transizione tra il bosco misto di alta quota e le praterie alpine, questo tipo di vegetazione sembra piuttosto rappresentare uno stadio della successione attraverso cui il bosco, un tempo eliminato dall'intervento antropico, sta gradualmente ricolonizzando le aree non più pascolate. La diffusione del mugo rappresenterebbe un primo stadio di tale ricolonizzazione. Tuttavia, anche se non rilevata, nell'area di studio è stata osservata su detriti calcarei pure l'associazione tipica descritta da Oberdorfer (1992).

Gruppo 8 (Tab. 6). Cembreta (*Vaccinio-Pinetum cembrae rhododendretosum hirsuti*).

I rilievi di questo gruppo sono localizzati nelle aree boscate circostanti i pascoli di Padeon caratterizzate da modeste pendenze e da terreni poco profondi. Essi formano un mosaico vegetazionale con le superfici interessate dal pascolo a *Sesleria albicans* (Gruppo 5). La composizione floristica è caratterizzata dalla presenza di specie di *Vaccinio-*

Piceetea, di *Erico-Pinetea* e da alcune specie della classe *Seslerietea albicantis*. Tale tipo di vegetazione sembra riferibile all'associazione *Vaccinio-Pinetum cembrae* ed in particolare alla subass. *rhododendretosum hirsuti* tipica dei substrati calcarei e caratterizzata dalla presenza di specie della classe *Erico-Pinetea* e di altre specie basifile (Mayer 1974, Oberdorfer 1992). I rilievi di questo gruppo sono floristicamente molto simili all'associazione *Calamagrostio villosae-Pinetum cembrae* descritta da Filipello *et al.* (1981) per le Alpi meridionali ed in particolare alla subass. *seslerietosum variae*, caratterizzata dalla presenza di numerose specie basifile. Questa associazione rappresenterebbe secondo gli autori succitati il tipo di cembreta più diffuso nelle Alpi orientali.

Tab. 6 - Cembreta (gruppo di ril. 8 di Fig. 3). Altitudine: 1840-1910 m; copertura della vegetazione: 100 %.

Pinus cembra community (relevé group 8 of Fig. 3).

Gruppo rilievi	8				PRE-SENZE
	69	6	9	2	
Numero rilievo	SO	SO	SO	SO	
Esposizione	9	14	14	12	
Inclinazione (%)	150	110	225	90	
Superficie di rilievo (mq)	24	29	35	24	
N. totale specie	29	18	15	13	
Profondità del suolo (cm)					
VACCINIO-PINETUM CEMBRAE					
RHODODENDRETOSUM HIRSUTI					
<i>Pinus cembra</i>	+	2	2	2	4
<i>Sorbus chamaemespilus</i>	1	2		1	3
<i>Rhododendron hirsutum</i>	2	1			2
<i>Pinus mugo</i>			+	3	2
RHODODENDRO-VACCINIENION					
<i>Clematis alpina</i>	1	+	+	+	4
<i>Juniperus communis</i> var. <i>montana</i>	2	2	3		3
PICEION ABIETIS					
<i>Calamagrostis villosa</i>	1	1	2	2	4
<i>Luzula sieberi</i> /sieberi	+	1	+	1	4
<i>Homogine alpina</i>	+	+	1	1	4
<i>Rhododendron ferrugineum</i>	3	1		1	3
<i>Luzula luzulina</i>			+		1
PICEETALIA ABIETIS e VACCINIO-PICEETEA					
<i>Picea excelsa</i>	2	2	2	2	4
<i>Vaccinium vitis-idaea</i>	2	2	3	2	4
<i>Vaccinium myrtillus</i>	2	2	2	1	4
<i>Pyrola rotundifolia</i>	1	1	1	+	4
<i>Erica carnea</i>	2	3	2	2	4
<i>Hieracium bifidum</i>	+	+	+	1	4
<i>Aposeris foetida</i>	1	1	1	1	4
<i>Melampyrum sylvaticum</i>		+	+	1	3
SPECIE COMPAGNE					
<i>Daphne striata</i>		2	1	1	3
<i>Sesleria albicans</i>	+	1	1		3
<i>Pedicularis elongata</i>	+	+	+		3
<i>Geranium sylvaticum</i>	r		r	1	3
<i>Campanula scheuchzeri</i>	+		+	+	3
<i>Valeriana montana</i>	1	1	+		3
<i>Potentilla erecta</i>	+		1	1	3
<i>Cirsium erisithales</i>			1	1	2
<i>Aster bellidiastrium</i>		+	1		2
<i>Carex ferruginea</i>		1	+		2
<i>Galium pumilum</i>		1	+		2
<i>Laserpitium peucedanoides</i>			+	1	2
<i>Solidago virga-aurea</i>	+		+		2
<i>Ranunculus montanus</i>			+	+	2
<i>Luzula nivea</i>	2		+		2

Valutazione ecologica dei tipi di vegetazione

In Tab. 7 sono riportate le medie ponderate degli indici ecologici di Landolt (1977) per i gruppi di rilievi vegetazionali precedentemente descritti. Sulla base di tale tabella è possibile ottenere una valutazione ecologica dei tipi vegetazionali individuati. L'indice di umidità presenta i valori più bassi nei gruppi 4c, 6 e 7 (2.1-2.4) relativi rispettivamente al seslerieto, al margine di ghiaione dolomitico ed alla mugheta. Si tratta di fitocenosi costituite principalmente da specie poco esigenti per ciò che riguarda il grado di umidità del suolo.

Il valore relativamente elevato (3.4) dell'indice per il gruppo 1 conferma il carattere notevolmente igrofilo del pascolo a *Poa alpina* di Padeon. I gruppi di rilievi 2 e 3 (formazioni a *Carex davalliana*) presentano i valori più elevati di tale indice (3.3-3.8).

L'indice di luce presenta il valore più basso (2.9) in corrispondenza della cembreta (gruppo 8), fitocenosi questa ricca di specie adattate a condizioni di scarsità di luce. I gruppi di rilievi 2, 3, 4a, 4b e 4c presentano i valori più elevati dell'indice (3.7-3.9), in quanto la loro composizione floristica è caratterizzata principalmente da specie di ambienti aperti.

L'indice di elementi nutritivi presenta la variabilità più elevata (2.1-3.8). Il valore più alto corrisponde al pascolo a *Poa alpina* (gruppo 1) per lo più per gli elevati valori di presenza e copertura di *Deschampsia caespitosa* (indice pari a 4). I valori più bassi si riscontrano nel seslerieto (gruppo 4c), nel margine del ghiaione dolomitico (gruppo 6), nella mugheta (gruppo 7) e nella cembreta (gruppo 8) (2.1-2.3). Per tale motivo sembra che in queste situazioni sia bassa la disponibilità di elementi nutritivi.

L'indice di humus varia entro limiti abbastanza ampi (2.8-4.2). Il valore più basso, relativo al margine del ghiaione dolomitico (gruppo 6), indica terreni con contenuto in humus da scarso a medio. Per contro, i valori più elevati, relativi alla mugheta (gruppo 7: 3.7) ed alla cembreta (gruppo 8: 4.2), indicano suoli con contenuto in humus buono o elevato.

L'indice di granulometria varia pure in un intervallo abbastanza ampio (3.0-4.3). Il valore minimo corrisponde alla vegetazione del margine del ghiaione dolomitico (gruppo 5) ed indica l'elevato grado di scioltezza del terreno di tale fitocenosi. I valori più alti (4.2-4.4) sono relativi alle formazioni a *Carex davalliana* (gruppi 2 e 3) ed al

pascolo a *Poa alpina* (gruppo 1) e indicano terreni con buona capacità di trattenuta idrica.

Tutti gli altri indici ecologici (temperatura, continentalità, reazione del suolo) presentano intervalli di variazione molto ristretti.

Tab. 7 - Medie ponderate degli indici ecologici (Landolt 1977) nei tipi di vegetazione di Malga Padeon individuati in base alla classificazione di Fig. 3.

Mean ecological indexes (Landolt 1977) of the vegetational types of Malga Padeon obtained with the classification of Fig. 3.

Gr. ril.	Umid.	Luce	Temp.	Cont.	Reaz.	Nutr.	Hum.	Gran.
1	3,4	3,3	2,9	2,9	3,1	3,8	3,3	4,3
2	3,8	3,9	2,6	2,7	3,6	2,5	3,5	4,4
3	3,3	3,7	2,6	2,9	3,2	2,7	3,5	4,2
4a	2,7	3,7	2,4	3,3	3,3	2,7	3,1	3,8
4b	2,5	3,8	2,3	3,5	3,4	2,3	3,0	4,0
4(a+b)	2,7	3,7	2,4	3,4	3,3	2,5	3,1	3,7
4c	2,4	3,7	2,6	3,4	3,2	2,3	3,0	3,5
5	2,7	3,4	2,3	3,1	3,3	2,3	3,4	3,6
6	2,1	3,1	2,8	3,1	3,3	2,1	2,8	3,0
7	2,3	3,4	2,3	3,7	3,1	2,2	3,7	3,6
8	2,6	2,9	2,4	3,2	2,8	2,2	4,2	3,8

Analisi della relazione fra micromorfologia del suolo e vegetazione

Nell'ambito del pascolo a *Carex ferruginea* (gruppi 4a e 4b) sono stati comparati quattro rilievi di dosso con quattro rilievi di avvallamento. La Tab. 8 riporta la composizione floristica di questi rilievi. In Tab. 8 le specie sono state raggruppate in tre gruppi: il gruppo 1 è maggiormente rappresentato sui dossi (d), il gruppo 2 comprende specie legate agli avvallamenti (a), il gruppo 3 è costituito da specie indifferenti alle condizioni micromorfologiche di dosso e di avvallamento.

Per ciò che riguarda il primo gruppo di specie, *Gentiana ciliata*, *Primula farinosa*, *Helianthemum oelandicum*, *Carlina acaulis* e *Selaginella selaginoides* risultano presenti quasi esclusivamente sui dossi mentre *Anthyllis vulneraria*, *Koeleria pyramidata*, *Sesleria albicans* e *Horminum pyrenaicum* presentano i valori di copertura più elevati sui dossi, pur essendo presenti anche negli avvallamenti. La gran parte delle specie del primo

Tab. 8 - Composizione floristica di rilievi di dosso (d) e di avvallamento (a); ulteriori spiegazioni nel testo.

Floristic composition of relevés located on rises (d) and in depressions (a); further explanations in the main text.

Gruppo di specie	Numero del rilievo	25 d	27 d	39 d	44 d	26 a	28 a	40 a	45 a
1	Plantago media	+	1	1	+	+	+	+	+
	Pedicularis elongata	+	1	1	1	+	+	1	+
	Trifolium pratense/nivale	1	1	+	+	+	+	+	+
	Anthyllis vulneraria	2	2	2	1	1	+	1	1
	Prunella grandiflora	2	2	2	2	+	+	1	+
	Lotus alpinus	1	+	1	1	+	+	+	+
	Potentilla crantzii	+	1	1	1	+	+	+	+
	Gentiana verna	+	1	1	1	+	+	+	+
	Sesleria albicans	2	2	2	1	+	+	+	+
	Horminum pyrenaicum	2	2	2	2	+	1	+	+
	Koeleria pyramidata	1	2	1	2	+		1	
	Helianthemum oelandicum	+	1	1	+	+			
	Selaginella selaginoides	+	+	+	+			+	
	Thymus alpigenus	2		1	+	+	+		
	Soldanella alpina	+	+	+				+	
	Erigeron polymorphus	1	+		+	+			
	Primula farinosa	+		1	1	+			
	Carlina acaulis	+	1	1	+				
	Gentiana ciliata	+	+		+				
	Carex ornithopoda	+		1					
	Aster bellidiastrum	+		1					
	Hippocrepis comosa	1		+					
2	Avenula pubescens	1	+	+	+	1	1	1	2
	Ranunculus acris	1	+		+	1	1	1	1
	Trollius europaeus	+		+	+	2	1	2	2
	Anthoxanthum alpinum	1		1		+	1	1	2
	Alchemilla gr.vulgaris	1			+	1	1	2	2
	Agrostis tenuis	1	+			+	2	1	1
	Aposeris foetida	+			+	2	+	1	
	Myosotis alpestris	+				+	+	+	1
	Achillea millefolium	+				1	1	1	2
	Poa alpina	1				+	+	+	1
	Deschampsia caespitosa	+				2	3	1	2
	Knautia longifolia				+	1	2	1	2
	Carex ferruginea	+		1		2		2	1
	Carex pallescens		+				1	+	+
	Senecio gaudinii					2	1	1	
	Festuca pratensis					2	1	+	
	Phleum alpinum					+	1	1	
	Dactylis glomerata	+				2	1		
	Rumex acetosa					+	+		+
3	Leucanthemum adustum	+	+	+	2	+	+	+	1
	Cardus carlineifolius	1	2	1	2	2	+	+	1
	Polygonum viviparum	+	+	+	+	+	+	+	+
	Carex sempervirens	2	2	2	2	2	2	2	2
	Scabiosa lucida	+	2	1	1	1	1	1	+
	Galium pumilum	1	+	1	+	+	+	+	+
	Briza media	+	2	2	2	2	+	2	2
	Luzula multiflora	+	+	+	+	+		+	+
	Carum carvi	+	+	1	+	+	1	+	
	Polygala alpestris	+		1	+	+	+	+	
	Campanula scheuchzeri	+		+	+	+	+	+	
	Phyteuma orbiculare	+		+	+	1	+	+	
	Rhinanthus freynii	2	1	1		1	+	1	
	Trifolium repens	+	+		+	+	+	+	
	Cirsium acaule	+		1	+		2		1
	Festuca valesiaca	2	2		2	2			2
	Homogine alpina		+	1	+	+		+	
	Ranunculus montanus	+		+	+		+	+	
	Crocus albiflorus	+	+			+	+	+	
	Potentilla erecta			+	+			+	+
	Silene vulgaris			+	+	1		1	
	Cerastium alpinum				+		+		+
	Festuca rubra			2			1	2	
	Veronica chamaedrys	+					+	+	
	Biscutella laevigata				+				+
	Thesium alpinum			+				+	

avvallamenti. Le specie di questo gruppo sono caratteristiche delle classi *Seslerietea albicantis* e *Festuco-Brometea*. Delle specie che appartengono al secondo gruppo *Senecio gaudinii*, *Festuca pratensis*, *Phleum alpinum* e *Rumex acetosa* sono presenti solamente nei rilievi di avvallamento, mentre le altre specie, presenti anche sui dossi, risultano particolarmente legate agli avvallamenti per i loro valori di presenza e copertura. Quasi tutte le specie di questo gruppo sono caratteristiche della classe *Molinio-Arrhenatheretea*.

Nel complesso il pascolo a *Carex ferruginea* in situazioni di dosso è caratterizzato da specie di seslerieto, in condizioni di avvallamento presenta entità del pascolo a *Poa alpina*. Una simile differenziazione della vegetazione in conseguenza della micromorfologia del suolo è stata descritta anche da Chiapella Feoli e Poldini (1993), che distinguono nell'ambito del fitocenon a *Horminum pyrenaicum* due sottotipi: uno a *Globularia cordifolia* (di dosso) e l'altro a *Trifolium pratense* (di avvallamento).

La Tab. 9 riporta i valori medi ponderati degli indici di Landolt (1977) per gli otto rilievi di dosso (d) e di avvallamento (a). Gli avvallamenti sembrano essere caratterizzati da suoli più ricchi in elementi nutritivi ed in humus, a granulometria più fine e più umidi.

Analisi indiretta di gradienti ecologici

La Fig. 5 riporta l'ordinamento dei rilievi nel piano individuati dalle due prime componenti principali. Tali componenti spiegano il 27,4% della varianza totale. La prima componente principale separa i rilievi di cembreta (gruppo 8) e di mugheta (gruppo 7) da quelli dei pascoli a *Carex ferruginea* (gruppi 4a e 4b). La seconda componente principale separa il pascolo a *Poa alpina* (gruppo 1) dai seslerieti (gruppi 4c e 5).

Dalle correlazioni fra gli indici di Landolt (1977) di Tab. 7 e i valori medi delle due prime componenti principali in ogni gruppo di rilievi (Fig. 5) risulta che i fattori luce, humus, elementi nutritivi e temperatura sono legati alla variazione floristica dei gruppi di rilievi ottenuti dalla classificazione. In particolare la prima componente principale è legata ai fattori luce e humus (Fig. 6), la seconda ai fattori elementi nutritivi e temperatura (Fig. 7).

Sulla base di tali risultati e di osservazioni di campo sono state ipotizzate le relazioni esistenti fra i diversi tipi vegetazionali rilevati (Fig. 8).

Tab. 9 - Medie ponderate degli indici ecologici (Landolt 1977) nei rilievi di Tab. 8; d: dosso; a: avvallamento.

Mean ecological indices (Landolt 1977) of the relevés of Tab. 8; d: rises; a: depressions.

N. ril.	Umid.	Luce	Temp.	Cont.	Reaz.	Nutr.	Hum.	Gran.
25 d	2,2	3,9	2,6	3,6	3,4	2,3	3,0	3,2
27 d	2,0	4,0	2,6	3,8	3,4	2,1	2,9	3,2
39 d	2,4	4,0	2,3	3,4	3,5	2,2	3,1	3,3
44 d	2,2	3,9	2,5	3,8	3,3	2,1	2,9	3,3
26 a	2,7	3,5	2,6	3,3	3,3	3,0	3,2	3,9
28 a	3,2	3,4	2,5	3,1	3,1	3,3	3,2	4,3
40 a	2,7	3,8	2,2	3,2	3,2	2,7	3,2	3,8
45 a	2,9	3,6	2,5	3,3	3,2	3,0	3,1	4,1
Media								
d	2,2	3,9	2,5	3,7	3,4	2,2	3,0	3,3
a	2,9	3,6	2,5	3,2	3,2	3,0	3,2	4,0

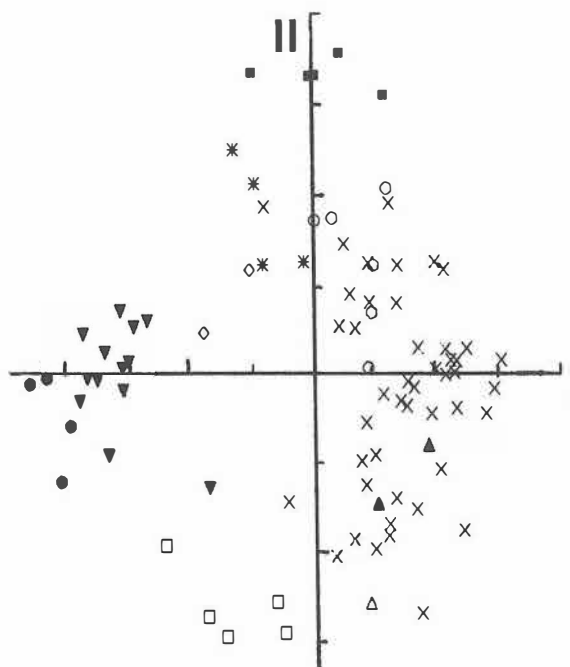


Fig. 5 - Ordination dei rilievi fitosociologici nel piano individuato dalle prime due componenti principali. Gruppi di rilievi come in Fig. 3; ■, 1; *, 2; ○, 3; ×, 4(a+b); ▼, 4c; □, 5; ◇, 6; ▲, 7; ●, 8. Ulteriori spiegazioni nel testo.

Ordination of the phytosociological relevés based on the two first components. Relevé groups as in Fig. 3; ■, 1; *, 2; ○, 3; ×, 4(a+b); ▼, 4c; □, 5; ◇, 6; ▲, 7; ●, 8. Further explanations in the main text.

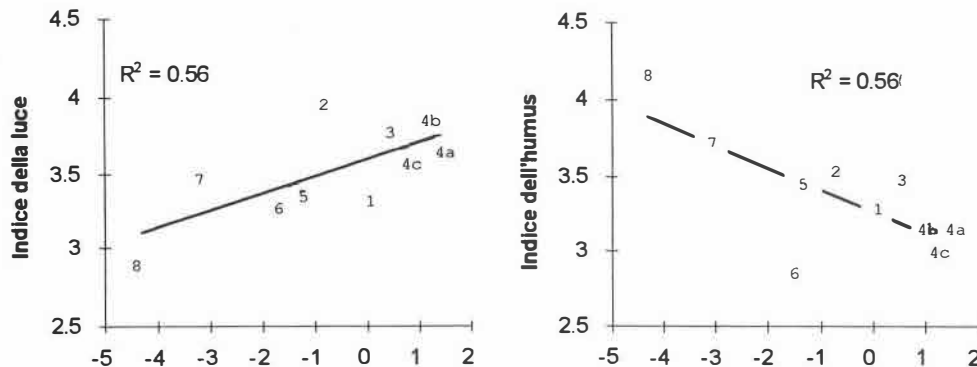


Fig. 6 - Correlazione tra la prima componente principale dell'ordinamento di Fig. 5 e i valori medi degli indici ecologici di luce e humus nei gruppi di rilievi (Tab. 7).

Correlation between the first principal component of the ordination of Fig. 5, and the mean values of the ecological indexes of light and humus in the relevé groups (Tab. 7).

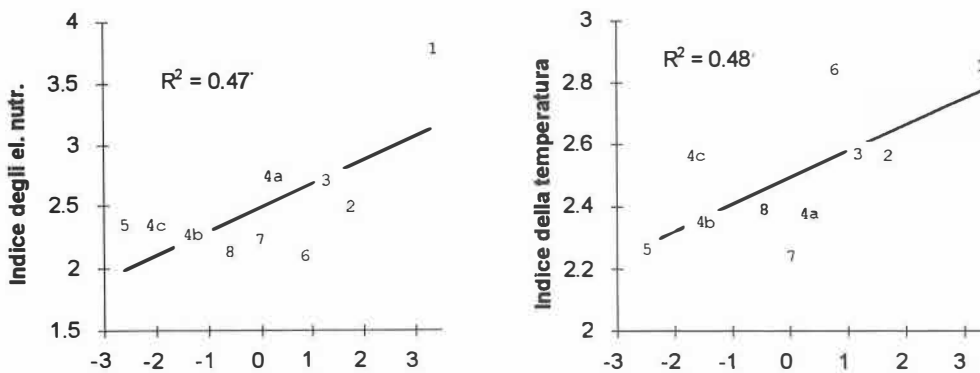


Fig. 7 - Correlazione tra la seconda componente principale dell'ordinamento di Fig. 5 e i valori medi degli indici ecologici di elementi nutritivi e di temperatura nei gruppi di rilievi (Tab. 7).

Correlation between the second principal component of the ordination of Fig. 5, and the mean values of the ecological indexes of nutritive elements and of temperature in the relevé groups (Tab. 7).

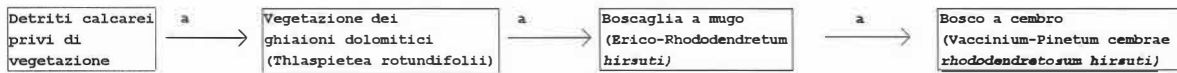
La serie dei tipi di vegetazione ghiaioni dolomitici - mugheta - cembreta sembra correlata all'aumento della profondità, degli elementi nutritivi e della capacità di trattenuta idrica dei suoli (Fig. 8, a).

L'attività alpicolturale sembra influire sul dinamismo sia con l'eliminazione delle specie arbustive ed arboree, sia con l'arricchimento del terreno in elementi nutritivi (Fig. 8, b e d). Il bosco a cembro (*Vaccinio-Pinetum cembrae rhododendretosum hirsuti*) viene in tal modo sostituito dal pascolo a *Carex ferruginea* (*Caricetum ferrugineae trifolietosum*) e, in seguito ad ulteriore apporto di elementi nutritivi (Fig. 8, c), dal pascolo a *Poa alpina* (*Poion alpinae*). Su terreni più superficiali si sviluppa il pascolo a *Sesleria albicans* (*Carici*

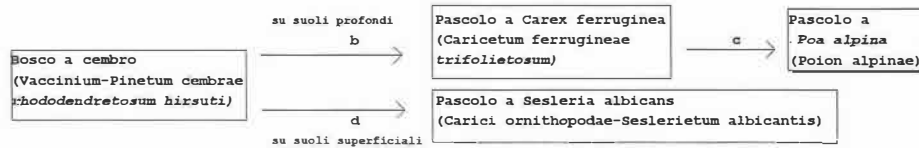
ornithopodae-Seslerietum albicantis).

In conseguenza dell'interruzione delle attività alpicolturali la ricostituzione della vegetazione forestale climax sembra avvenire attraverso uno stadio di ricolonizzazione a mugo (*Erico-Rhododendretum hirsuti*) (Fig. 8, sia sui terreni più profondi e fertili dei pascoli a *Poa alpina* (*Poion alpinae*) e a *Carex ferruginea* (*Caricetum ferrugineae trifolietosum*) sia sui terreni relativamente superficiali del pascolo a *Sesleria albicans* (*Carici ornithopodae-Seslerietum albicantis*). La ricostituzione della vegetazione forestale è seguita da una riduzione del contenuto in elementi nutritivi del suolo e da una riduzione del fattore luce. Un'altra probabile via di ricostituzione consiste nel passaggio diretto dal pascolo a *Sesleria*

In assenza di intervento antropico



In presenza di intervento antropico



In seguito all'abbandono del pascolo

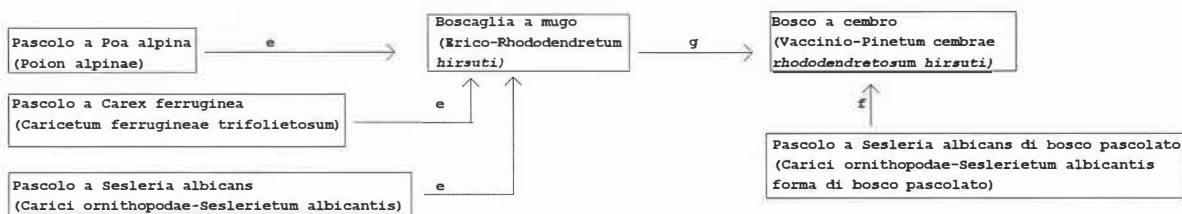


Fig. 8 - Schema delle relazioni fra i diversi tipi di vegetazione individuati; a: aumento della capacità idrica e del contenuto in elementi nutritivi del suolo; b: taglio di alberi ed arbusti e pascolamento con moderato apporto di elementi nutritivi; c: aumento dell'intensità del pascolamento con ulteriore apporto di elementi nutritivi; d: taglio di alberi ed arbusti e pascolamento con scarso apporto di elementi nutritivi; e: interruzione del pascolamento ed invasione di *Pinus mugo*; f: interruzione del pascolamento ed insediamento di piante arbustive ed arboree; g: sostituzione di *Pinus mugo* con piante arboree.

Scheme of the relationships among the vegetational types; a: increase in soil water capacity and nutrient content; b: cut of trees and scrubs and grazing with moderate supply of nutrients; c: increase in grazing pressure with additional supply of nutrients; d: cut of trees and scrubs and grazing with low supply of nutrients; e: interruption of grazing and recolonization of *Pinus mugo*; f: interruption of grazing and recolonization of trees and scrubs; g: replacement of *Pinus mugo* by trees.

albicans (*Carici ornithopodae-Seslerietum albicantis*) di bosco pascolato al bosco a cembro (*Vaccinio-Pinetum cembrae rhododendretosum hirsuti*) (Fig. 8, f). Con la progressiva chiusura dello strato arboreo a causa della cessazione dei tagli (Fig. 8, g) il mugo, specie a carattere eliofilo, sembra regredire, e vengono favoriti soprattutto i mirtilli e le specie arboree caratteristiche della vegetazione climax (cembro, larice ed abete rosso).

Valutazione agronomica dei pascoli

In Fig. 9 è riportato il risultato della classificazione automatica dei 28 rilievi lineari di pascolo. Tale figura evidenzia sei gruppi di rilievi tipizzati come ecofacies. In Tab. 10 è riportata per ogni ecofacies individuata il contributo specifico medio delle specie più frequenti ed il suo valore foraggero medio. La distribuzione sull'area indagata delle diverse ecofacies di pascolo individuate è

riportata nella mappa di Fig.10.

Le caratteristiche vegetazionali e la valutazione agronomica delle ecofacies individuate (Tab. 10) sono le seguenti:

a) Ecofacies a *Deschampsia caespitosa*, *Alchemilla vulgaris* e *Achillea millefolium*

Questa ecofacies risulta caratterizzata principalmente da elevati valori di contributo specifico di *Deschampsia caespitosa*, *Alchemilla vulgaris* e *Achillea millefolium* (Tab. 10, a). Delle sei ecofacies individuate quella in esame possiede il più alto valore foraggero medio (48.0), nonostante l'abbondanza di *Deschampsia caespitosa*, che presenta un interesse foraggero relativamente basso. La produzione media annua di sostanza secca rilevata in questa ecofacies è pari a 1.89 t/ha (2.03 t/ha nel 1991 e 1.74 t/ha nel 1992). Dei quattro rilievi di questa ecofacies tre (17, 32 e 41) rappresentano la parte centrale del pascolo,

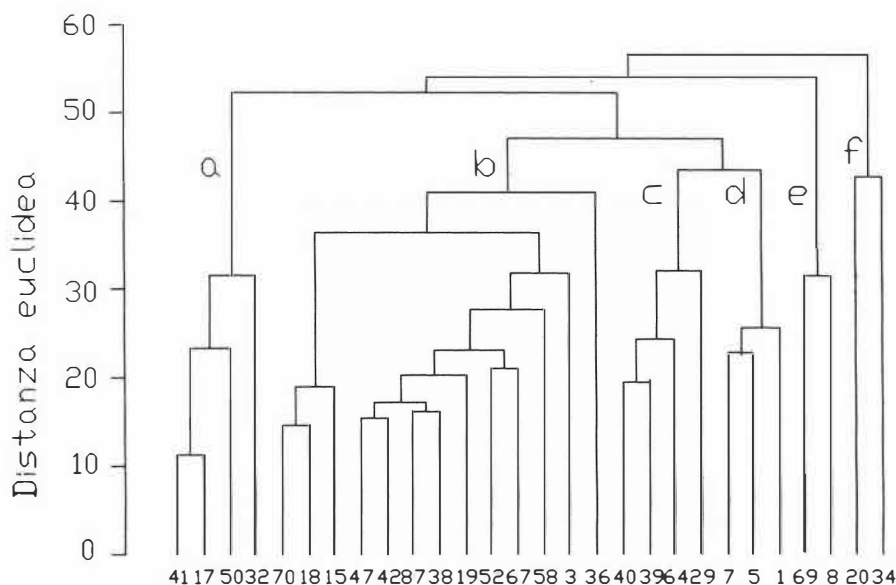


Fig. 9 - Dendrogramma dei rilievi lineari dei pascoli.; le lettere indicano le ecofacies descritte nel testo.

Dendrogram of the linear relevés of pastures; groups indicated by letters correspond to the ecofacies discussed in the main text.

caratterizzata da terreno profondo e probabilmente più fertile. Tali rilievi, oltre alle specie citate, presentano contributi specifici relativamente elevati di alcune specie buone foraggere tra cui *Dactylis glomerata*, *Veronica chamaedrys* e *Festuca pratensis*. Il quarto rilievo (50) presenta invece il massimo valore di contributo specifico di *Deschampsia caespitosa* (32.0), alla quale si accompagnano altre entità di scarso interesse foraggero quali *Briza media*, *Polygonum viviparum*, *Galium pumilum* e *Sesleria albicans*.

b) Ecofacies a *Briza media* e *Polygonum viviparum*

Questa ecofacies costituisce la gran parte del pascolo di Padeon e si sviluppa in aree più o meno pianeggianti con terreno mediamente profondo. Il valore foraggero medio (31.0) risulta inferiore rispetto a quello della precedente ecofacies. Ciò è dovuto, innanzitutto, alla presenza di numerose specie di valore foraggero basso (*Sesleria albicans*, *Carex sempervirens* e *Prunella grandiflora*), nullo (*Polygonum viviparum*) oppure di specie velenose (*Ranunculus acris*). Le specie buone foraggere (soprattutto leguminose) contribuiscono in misura ridotta alla composizione del pascolo. La produzione media annua di sostanza secca è di 1.8 t/ha (1.93 t/ha nel 1991 e 1.67 t/ha nel 1992). In base alla composizione floristica è possibile suddividere questa ecofacies nelle seguenti quattro subfacies:

1) Subfacies a *Anthoxanthum alpinum* e *Festuca rubra*: valore foraggero medio 32.2.

2) Subfacies a *Carex sempervirens* e *Sesleria albicans*: costituisce l'aspetto più comune dell'ecofacies b). Il valore foraggero medio è di 29.1.

3) Subfacies a *Sesleria albicans* e *Leontodon hispidus* (ril. 3): si sviluppa su suolo relativamente superficiale e asciutto. Il valore foraggero medio (34.8) è alto soprattutto per la presenza di *Leontodon hispidus*, specie di elevata qualità foraggera.

4) Subfacies a *Willemetia stipitata* (ril. 36): si sviluppa su suolo molto umido e profondo. Nonostante la consistente presenza di alcune specie di basso interesse pabulare (*Carex davalliana* e *Nardus stricta*) il valore foraggero medio di questa subfacies è elevato per l'abbondanza di *Willemetia stipitata*, specie dalle caratteristiche foraggere simili a quelle di *Leontodon hispidus*.

c) Ecofacies a *Festuca rubra*, *Carex sempervirens* e *Briza media*

Questa ecofacies è tipica di aree con suoli poco profondi della fascia di transizione tra il pascolo vero e proprio e il bosco pascolato. Essa presenta notevole affinità floristica con l'ecofacies b) soprattutto per l'abbondanza di *Briza media* e *Carex sempervirens*, ma da questa si differenzia per il maggior contributo specifico di alcune specie relativamente xerofile e di *Festuca rubra*. Sia la

Tab. 10 - Copertura percentuale media delle specie e valore foraggero medio nelle ecofacies di pascolo individuate in base alla classificazione dei rilievi lineari di Fig. 9; sono riportate solo le specie con copertura superiore all'1% in almeno una ecofacies. Ulteriori spiegazioni nel testo.

Species mean per cent cover and mean fodder value in the pasture ecofacies resulting from the linear relevés classification of Fig. 9; species with a cover higher than 1% are reported. Further explanations in the main text.

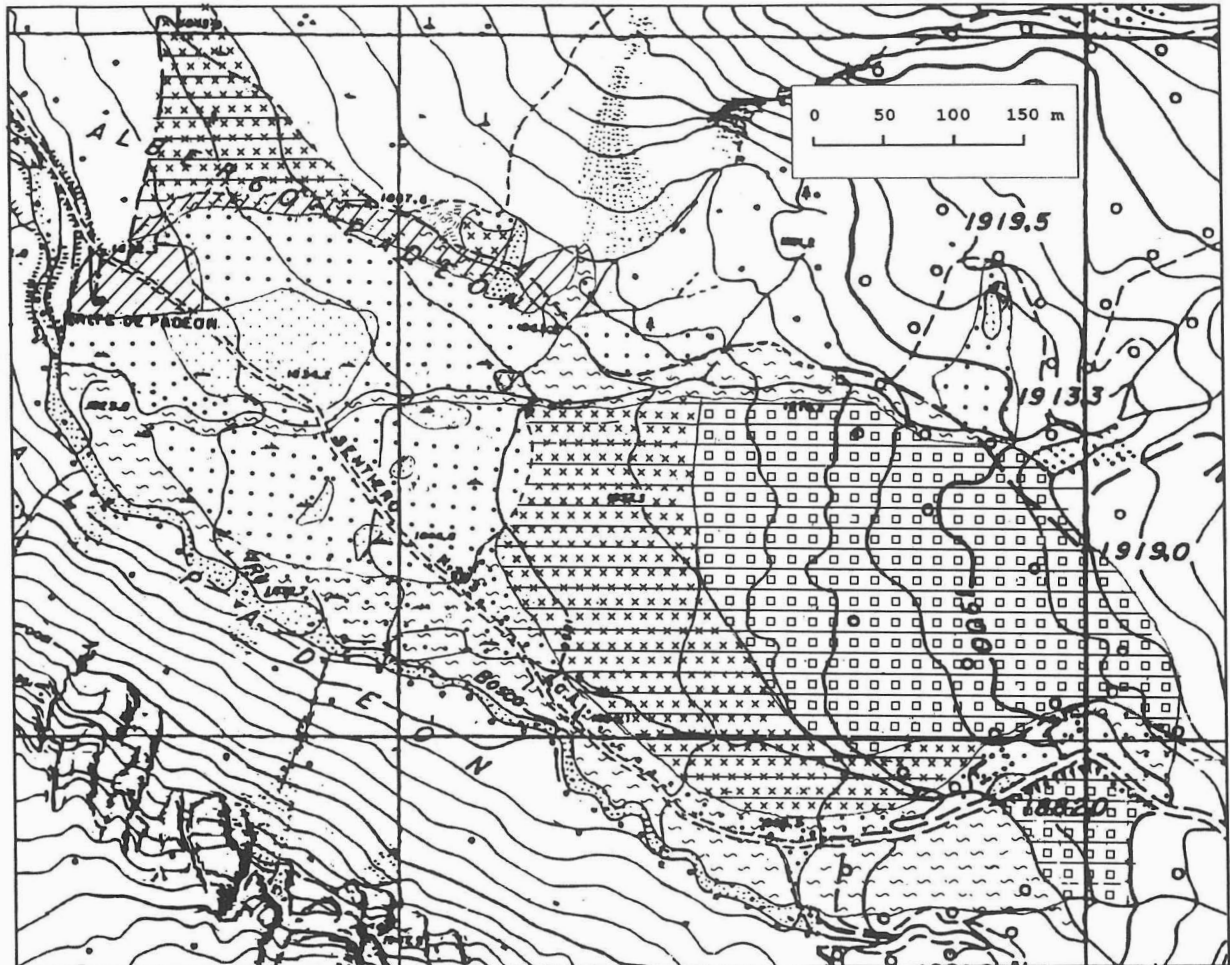
Valore foragg. specie	Ecofacies Subfacies Valore foraggero medio	a	1	2	3	4	Media	c	d	e	f
		48,0	32,2	29,1	34,8	39,4	31,0	30,3	14,7	2,2	18,1
3	<i>Deschampsia caespitosa</i>	25,6	2,7	2,0		3,6	2,1				
5	<i>Alchemilla gr.vulgaris</i>	11,6	2,7			2,4	0,8				3,8
5	<i>Achillea millefolium</i>	8,4	4,4	1,4			1,9				
-1	<i>Senecio gaudinii</i>	2,3									
-1	<i>Ranunculus acris</i>	3,6	5,3	1,4			2,1				
0	<i>Polygonum viviparum</i>	1,3	6,7	5,3	1,4	4,2	5,3	2,0	1,2		1,8
5	<i>Briza media</i>	1,3	4,3	10,1	8,7	15,2	9,1	9,9			1,2
5	<i>Festuca rubra</i>	8,8	7,6	3,7			4,0	17,4	7,9		
0	<i>Carex sempervirens</i>		2,7	12,2	1,1	1,8	8,3	11,7	11,3		
2	<i>Sesleria albicans</i>			9,9	18,1		7,5	7,8	19,4	3,1	
1	<i>Carex ferruginea</i>			2,0		4,5	1,6	1,3	2,6	3,5	2,9
1	<i>Horminum pyrenaicum</i>		3,7	5,0	7,9		4,5	6,6	2,4		
2	<i>Potentilla erecta</i>				1,8	2,1	0,3	2,2	1,6		
2	<i>Prunella grandiflora</i>		4,6	7,5			5,7	3,6			
0	<i>Thymus alpinus</i>		2,0	1,1			1,2	2,7			
7	<i>Lotus alpinus</i>			3,4	2,5		2,3	2,3			1,7
-1	<i>Trollius europaeus</i>		1,1	1,2			1,0	2,1			
5	<i>Leontodon hispidus</i>	1,6		2,4	12,6		2,4				
7	<i>Trifolium pratense/nivale</i>		3,0	1,3		1,2	1,6				1,7
3	<i>Festuca valesiaca</i>			1,5	10,8		1,8				
3	<i>Anthoxanthum alpinum</i>		12,9				3,0				
0	<i>Crocus albiflorus</i>		4,1				0,9				
0	<i>Carex ericetorum</i>				3,6		0,3				
6	<i>Willemetia stipitata</i>					20,8	1,6				
2	<i>Nardus stricta</i>					8,9	0,7				
5	<i>Hippocrepis comosa</i>				6,5		0,5	1,6			
0	<i>Leucanthemum adustum</i>				1,1		0,1	1,4			
-1	<i>Ranunculus montanus</i>					3,3	0,3	1,2			1,7
0	<i>Carlina acaulis</i>							2,9			
5	<i>Agrostis tenuis</i>							1,7			
1	<i>Hieracium glaciale</i>							1,2			
1	<i>Aposeris foetida</i>				1,8		0,1	2,0	8,7	1,9	
2	<i>Luzula sieberi</i>								6,3		
-1	<i>Melampyrum sylvaticum</i>								5,3		
1	<i>Homogone alpina</i>								3,5		
0	<i>Calamagrostis villosa</i>								3,4	3,0	
0	<i>Erica carnea</i>								5,4	19,8	
0	<i>Vaccinium vitis-idaea</i>								8,8	17,4	
0	<i>Juniperus communis</i>									15,0	
0	<i>Rhododendron ferrugineum</i>									7,4	
0	<i>Rhododendron hirsutum</i>									5,4	
0	<i>Vaccinium myrtillus</i>									8,2	
1	<i>Carex davalliana</i>					11,0	0,8				19,3
2	<i>Carex panicea</i>					4,8	0,4				16,5
0	<i>Cratoneuron commutatum</i>										16,8
1	<i>Juncus alpino-articulatus</i>										12,9
1	<i>Carex fusca</i>										3,8













produzione media annua di sostanza secca (1.8 t/ha; 1.99 t/ha nel 1991 e 1.61 t/ha 1992) che il valore foraggero medio (30.3) di questa ecofacies sono simili a quelli dell'ecofacies b).

d) Ecofacies a *Sesleria albicans*, *Carex sempervirens* e *Vaccinium vitis-idaea*

I rilievi di questa ecofacies rappresentano la vegetazione prevalentemente erbacea che si sviluppa su suolo relativamente profondo nel bosco pascolato. Dal punto di vista floristico questa ecofacies

costituisce una forma di transizione tra le ecofacies c) ed e). Infatti, accanto a entità tipicamente pabulari come *Carex sempervirens*, *Sesleria albicans* e *Festuca rubra*, sono presenti con elevati valori di contributo specifico *Vaccinium vitis-idaea* ed altre specie nemorali quali *Luzula sieberi*, *Melampyrum sylvaticum* e *Aposeris foetida*. Il valore foraggero medio (14.7) risulta di conseguenza intermedio rispetto a quelli delle ecofacies c) ed e), mantenendosi comunque ad un livello ancora apprezzabile soprattutto per l'elevato contributo di *Festuca rubra*.



- | | | | |
|---|--|---|--|
|  | Ecofacies a <i>Deschampsia caespitosa</i> , <i>Alchemilla vulgaris</i> e <i>Achillea millefolium</i> (a) 1.1 ha |  | Mosaico tra ecofacies d ed ecofacies a <i>Pinus mugo</i> e <i>Pinus cembra</i> * 5 ha |
|  | Ecofacies a <i>Briza media</i> e <i>Polygonum viviparum</i> (b) 5.6 ha |  | Ecofacies a <i>Carex davalliana</i> , <i>Carex panicea</i> e <i>Juncus alpino-articulatus</i> (f) 0.1 ha |
|  | Mosaico tra ecofacies b ed ecofacies a <i>Pinus mugo</i> * 0.2 ha |  | Ecofacies a <i>Pinus mugo</i> * 4.5 ha |
|  | Ecofacies a <i>Festuca rubra</i> , <i>Carex sempervirens</i> e <i>Briza media</i> (c) *0.6 ha |  | Ecofacies a <i>Pinus mugo</i> e <i>Pinus cembra</i> * 0.1 ha |
|  | Mosaico tra ecofacies c ed ecofacies a <i>Pinus mugo</i> * 0.2 ha |  | Ecofacies a <i>Picea excelsa</i> * 0.1 ha |
|  | Mosaico tra ecofacies a <i>Sesleria varia</i> , <i>Carex sempervirens</i> e <i>Vaccinium vitis-idaea</i> (d), ecofacies a <i>Erica carnea</i> , <i>Vaccinium vitis-idaea</i> e <i>Juniperus communis</i> (e) ed ecofacies a <i>Pinus mugo</i> e <i>Pinus cembra</i> * 7.3 ha |  | Ecofacies a <i>Silene vulgaris</i> * 0.1 ha |

* Ecofacies ottenute sulla base dei rilievi fitosociologici.

Fig. 10 - Mappa delle ecofacies di Malga Padeon.
Ecofacies map of Malga Padeon.

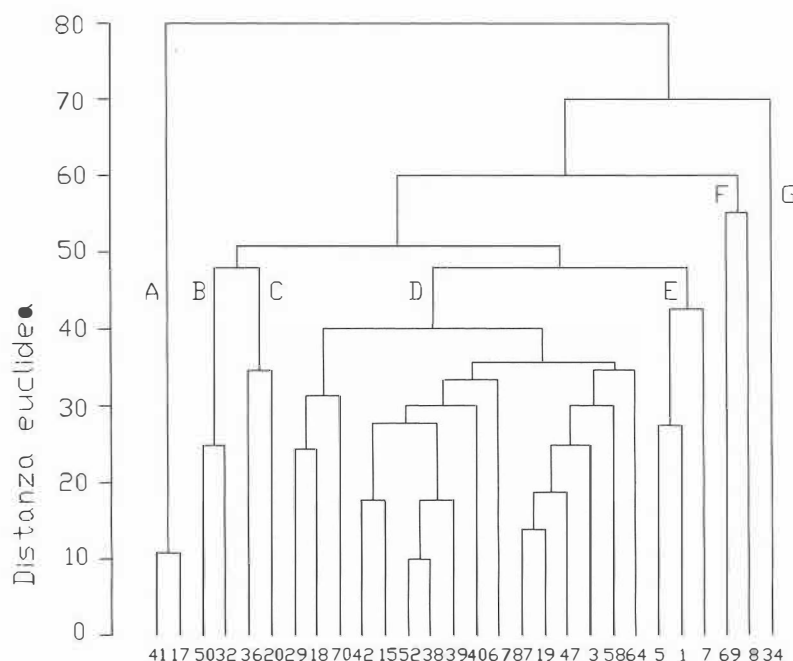


Fig. 11 - Dendrogramma dei rilievi fitosociologici dei pascoli; le lettere indicano le ecofacies come in Tab. 11.
Dendrogram of the phytosociological relevés of pastures; groups indicated by letters as in Tab. 11.

e) Ecofacies a *Erica carnea*, *Vaccinium vitis-idaea* e *Juniperus communis*

I rilievi di questa ecofacies rappresentano la vegetazione arbustiva che si alterna a mosaico con l'ecofacies d) all'interno del bosco pascolato su suoli abbastanza profondi. Il valore foraggero delle tre specie arbustive che ne dominano la composizione floristica (*Erica carnea*, *Vaccinium vitis-idaea* e *Juniperus communis*) è nullo e le specie erbacee pabulari sono presenti in scarsa quantità. L'interesse agronomico risulta, pertanto, nullo (valore foraggero medio = 2.2).

f) Ecofacies a *Carex davalliana*, *C. panicea* e *Juncus alpino-articulatus*

Questa ecofacies si sviluppa su superfici più o meno pianeggianti con terreni ad elevato contenuto idrico. Le specie che la caratterizzano, fra cui molte appartenenti al genere *Carex*, possiedono un valore foraggero basso. Tuttavia, alcune specie buone foraggiere quali *Lotus alpinus*, *Alchemilla vulgaris*, *Poa alpina* e *Trifolium pratense* pur presenti in quantità ridotta contribuiscono a mantenere ad un livello accettabile il valore foraggero medio dell'ecofacies (18.1). Comunque tale ecofacies presenta un interesse agronomico marginale in quanto occupa nell'area una superficie limitata.

Influenza del tipo di campionamento sulla valutazione agronomica

La Fig. 11 riporta il dendrogramma della classificazione automatica dei 28 rilievi fitosociologici eseguiti sulle medesime aree dei rilievi lineari. Tale classificazione ha evidenziato sette ecofacies di cui in Tab. 11 si riportano le denominazioni, i valori foraggeri medi e, in modo semplificato, la composizione floristica.

In Tab. 12 è riportata la tabella di contingenza dei rilievi nelle ecofacies individuate in base ai rilievi lineari (righe) ed ai rilievi fitosociologici (colonne). La struttura di Tab. 12 ed il coefficiente di correlazione cofenetica tra i dendrogrammi di Fig. 9 e Fig. 11 (0.445) indicano una buona corrispondenza fra le ecofacies individuate con i due diversi metodi di campionamento.

I valori foraggeri dei rilievi ottenuti in base ai due metodi di rilevamento sono riportati in Tab. 13. Il coefficiente di correlazione fra le due serie di valori è pari a 0.9. E' interessante notare che i valori foraggeri medi delle ecofacies individuate con l'analisi lineare calcolati in base ai contributi specifici risultano molto simili a quelli calcolati in base ai valori di abbondanza-dominanza dei rilievi fitosociologici (Tab. 13).

Tab. 11 - Copertura percentuale media delle specie e valore foraggero medio nelle ecofacies di pascolo individuate in base alla classificazione dei rilievi fitosociologici di Fig. 11; sono riportate solo le specie con copertura superiore all'1%.

Species mean percentage cover and mean fodder value in the pasture ecofacies resulting from the phytosociological relevés classification of Fig. 11: only species with a cover higher than 1% are reported.

A: ecofacies a *Deschampsia caespitosa* e *Alchemilla vulgaris*; B: ecofacies a *Deschampsia caespitosa*, *Alchemilla vulgaris* e *Festuca pratensis*; C: ecofacies a *Carex davalliana* e *Willemetia stipitata*; D: ecofacies a *Carex sempervirens*, *Horminum pyrenaicum* e *Sesleria albicans*; E: ecofacies ad *Aposeris foetida* e *Sesleria albicans*; F: ecofacies a *Juniperus communis*, *Rhododendron ferrugineum*, *Erica carnea* e *Picea excelsa*; G: ecofacies a *Cratoneuron commutatum*.

Valore foragg. specie	Ecofacies (1)	A	B	C	D	E	F	G
	Valore foraggero medio	41,9	43,3	27,7	27,0	12,1	1,4	12,8
3	<i>Deschampsia caespitosa</i>	62,0	23,5	2,6	1,9			3,0
5	<i>Alchemilla gr. vulgaris</i>	7,5	6,5	3,6	1,3			3,0
8	<i>Festuca pratensis</i>	1,5	7,5		0,1			
-1	<i>Ranunculus acris</i>	1,5	4,5	1,6	0,9	0,5	0,5	1,0
2	<i>Veronica chamaedris</i>	5,0	0,6	0,1	0,1			
7	<i>Dactylis glomerata</i>	4,0	2,0					
5	<i>Agrostis tenuis</i>		6,5	0,2	1,0			
-1	<i>Rhinanthus freynii</i>	0,6	4,0	1,1	2,5	0,1		0,2
-1	<i>Senecio gaudinii</i>	0,5	3,5		0,5			
0	<i>Carex flacca</i>		3,5	1,1	1,7			
7	<i>Trifolium pratense/nivale</i>	0,2	3,0	1,5	1,4			0,2
7	<i>Taraxacum alpinum</i>	0,6	3,0	0,1				
5	<i>Briza media</i>	0,5	3,0	6,0	4,6			0,2
5	<i>Festuca rubra</i>	0,2	2,0	0,1	4,9	1,5	0,1	2,0
0	<i>Carex sempervirens</i>	0,1	1,1	0,6	11,8	4,0	0,5	
1	<i>Horminum pyrenaicum</i>		1,0		8,7	7,1		
2	<i>Sesleria albicans</i>		0,5	0,2	7,5	11,3	1,6	
1	<i>Carex ferruginea</i>		0,1	6,5	3,6	10,7	2,0	0,2
2	<i>Prunella grandiflora</i>	0,5	0,1	1,1	3,4	0,1		
5	<i>Anthyllis vulneraria</i>			0,6	3,4	0,1		
3	<i>Anthoxanthum alpinum</i>		1,0	1,0	3,0	0,4		
1	<i>Aposeris foetida</i>		0,1	0,5	0,5	13,3	2,0	0,2
2	<i>Luzula sieberi</i>					5,7	1,1	
0	<i>Calamagrostis villosa</i>					5,3	1,5	
0	<i>Juniperus communis</i>					1,1	22,5	
0	<i>Rhododendron ferrugineum</i>						17,5	
0	<i>Erica carnea</i>					3,4	15,0	
0	<i>Picea excelsa</i>				0,3		15,0	
0	<i>Vaccinium vitis-idaea</i>					6,3	9,5	
0	<i>Rhododendron hirsutum</i>						7,5	
0	<i>Vaccinium myrtillus</i>			0,1		3,1	6,0	
1	<i>Carex davalliana</i>			14,0				10,0
6	<i>Willemetia stipitata</i>			10,0				
2	<i>Carex panicea</i>			7,5				6,0
0	<i>Carex paniculata</i>			4,0				
0	<i>Cratoneuron commutatum</i>			1,5				50,0
1	<i>Carex fusca</i>			5,0				5,0
1	<i>Juncus alpino-articulatus</i>			5,0				0,2
0	<i>Petasites albus</i>			3,1	1,4			2,0

Tab. 12 - Tabella di contingenza dei rilievi nelle ecofacies individuate in base alla classificazione dei rilievi lineari (righe) ed in base ai rilievi fitosociologici (colonne). Sigle delle ecofacies come in Figg. 9 e 11.

Contingency table of the relevés in the ecofacies singled out by the classification of the linear relevés (rows) and of the phytosociological relevés (columns). Letters as in Figs. 9 and 11.

	A	B	C	D	E	F	G
a	2	2	-	-	-	-	-
b	-	-	1	12	-	-	-
c	-	-	-	4	-	-	-
d	-	-	-	-	3	-	-
e	-	-	-	-	-	2	-
f	-	-	1	-	-	-	1

Tab. 13 - Valori foraggeri dei rilievi lineari e dei rilievi fitosociologici; ecofacies e subfacies come in Tab. 10.

Fodder values of the linear relevés and of the phytosociological relevés; ecofacies and subfacies as in Tab. 10.

Ecofacies	N. ril.	ril. lineare	ril. fitosoc.
a	41	51	41
	17	48	43
	50	39	37
	32	54	49
media		48	43
b1	70	27	21
	18	37	21
	15	32	23
media		32	22
b2	47	29	32
	42	26	27
	87	28	24
	38	31	34
	19	29	21
	52	30	38
	67	30	27
	58	30	33
media		29	29
b3	3	35	26
b4	36	39	40
c	40	32	32
	39	32	31
	64	32	23
	29	25	18
media		30	26
d	7	13	15
	5	13	13
	1	19	9
media		15	12
e	69	1	1
	8	4	2
media		2	1
f	20	22	16
	34	14	13
media		18	14

Conclusioni

Lo studio condotto nell'area di Malga Padeon ha posto in evidenza la presenza di pascoli di grande interesse pabulare e vegetazionale.

Grazie alla fertilità naturale dei suoli i valori foraggeri medi dei tipi di vegetazione individuati sono risultati relativamente elevati e comunque mediamente superiori a quelli che si riscontrano in pascoli di quote simili a trattamento semiestensivo. L'analisi fitosociologica ha evidenziato una notevole variabilità vegetazionale e una grande ricchezza floristica dovute principalmente alle modalità semiestensive con le quali in passato venne esercitata l'attività apicolturale.

Inoltre, degna di nota è la presenza nell'area di studio di fitocenosi ascrivibili all'associazione *Caricetum ferrugineae*, un *syntaxon* raramente segnalato e rilevato sulle Alpi orientali italiane.

In seguito all'interruzione del pascolamento sembra avere un ruolo di primo piano il mugo che, spesso presente ai bordi delle superfici a copertura erbacea un tempo pascolate, è in grado di espandersi all'interno di queste. Dal punto di vista floristico, la muggheta è il tipo di vegetazione fra quelli indagati più povero di specie e quindi una sua espansione nei pascoli abbandonati comporterebbe una perdita di biodiversità.

Tali osservazioni evidenziano come sia fortemente opportuno che la pianificazione territoriale competente, basata su un concetto di salvaguardia attiva oltretutto sulla necessità del divieto, preveda interventi atti a conservare i pascoli di Padeon, tanto più che gli stessi sono inseriti in un parco di grande interesse naturalistico quale quello delle Dolomiti d'Ampezzo.

Appendice

Specie sporadiche di Tab. 1 - Ril. 32: *Carex pallescens* e *Geum rivale*. Ril. 14: *Cerastium arvense*, *Daphne mezereum*, *Galeopsis speciosa*, *Erigeron acer*, *Erigeron polymorphus*, *Lamium album*, *Lathyrus pratensis* e *Scabiosa lucida*. Ril. 16: *Cerastium alpinum* subsp. *alpinum*, *Festuca valesiaca*, *Gentiana verna*, *Glechoma hederacea*, *Glyceria maxima*, *Pedicularis elongata*, *Polygala alpestris*, *Veratrum album* e *Viola tricolor*. Ril. 17: *Carex atrata*. Ril. 41: *Prunella grandiflora* e *Stellaria graminea*.

Specie sporadiche di Tab. 2 - Ril. 84: *Camptothecium lutescens* e *Crocus albiflorus*. Ril. 11: *Colchicum autumnale*. Ril. 85: *Agrostis alpina*, *Knautia longifolia*, *Pinus mugo* e *Tortella tortuosa*. Ril. 34: *Agrostis stolonifera*, *Arabis soyeri*, *Cratoneurum commutatum*, *Epilobium alsinifolium*, *Salix appendiculata*, *Salix daphnoides*, *Salix purpurea*, *Triglochin palustre*, *Valeriana dioica* e *Viola biflora*. Ril. 36: *Chaerophyllum hirsutum*, *Geum montanum*,

Koeleria pyramidata, *Lychnis flos-cuculi* e *Valeriana dioica*. Ril. 4: *Aster bellidiflorus*, *Carex ornithopoda*, *Erica carnea*, *Homogone alpina*, *Leontodon autumnalis*, *Orchis maculata*, *Potentilla crantzii*, *Thymus alpinus* e *Vaccinium myrtillus*. Ril. 22: *Carlina acaulis*, *Centaurea nervosa*, *Dactylis glomerata*, *Daphne mezereum*, *Festuca pratensis*, *Juniperus communis* var. *montana*, *Koeleria pyramidata*, *Peucedanum ostruthium*, *Pinus mugo*, *Rumex acetosa*, *Urtica dioica* e *Vicia sepium*. Ril. 20: *Agrostis stolonifera*, *Aster bellidiflorus*, *Brachytecium rivulare*, *Brachytecium salebrosum*, *Bryum turbinatum*, *Carex paniculata*, *Hieracium bifidum*, *Hypnum cupressiforme*, *Lychnis flos-cuculi*, *Mnium punctatum*, *Philonotis fontana*, *Silene vulgaris*, *Tortella tortuosa* e *Vaccinium myrtillus*. Ril. 21: *Achillea millefolium*, *Aconitum lamarkii*, *Anthriscus sylvestris*, *Carduus nutans*, *Carex ornithopoda*, *Cirsium acaule*, *Erica carnea*, *Euphrasia pulchella*, *Fragaria vesca*, *Hieracium piloselloides*, *Orobancha gracilis*, *Stachys alopecurus* e *Thymus alpinus*. Ril. 83: *Carduus nutans*, *Crocus albiflorus*, *Festuca pratensis* e *Veratrum album*.

Specie sporadiche di Tab. 3 - Ril. 37: *Centaurea nervosa* e *Veratrum album*. Ril. 33: *Gentiana ciliata* e *Veratrum album*. Ril. 28: *Aconitum lamarkii*, *Carex lepidocarpa*, *Galium mollugo* e *Geum montanum*. Ril. 15: *Aconitum lamarkii*, *Cerastium arvense*, *Galium mollugo* e *Hypericum maculatum*. Ril. 25: *Chenopodium bonus-henricus*, *Gentiana ciliata*, *Hieracium glaciale*, *Silene nutans* e *Trifolium badii*. Ril. 70: *Stellaria graminea*. Ril. 47: *Luzula spicata*, *Potentilla aurea*, *Pseudoleskea incurvata* e *Stellaria graminea*. Ril. 50: *Cerastium arvense*. Ril. 82: *Carex ericetorum* e *Parnassia palustris*. Ril. 80: *Parnassia palustris*, *Potentilla aurea* e *Veratrum album*. Ril. 66: *Arabis soyeri*, *Centaurea nervosa*, *Helianthemum nummularium* e *Scorzonera aristata*. Ril. 79: *Stellaria graminea*. Ril. 27: *Gentiana ciliata*. Ril. 52: *Centaurea nervosa*, *Daphne striata*, *Hieracium piloselloides*, *Melampyrum sylvaticum*, *Orobancha gracilis* e *Trifolium badii*. Ril. 43: *Hieracium glaciale* e *Laserpitium peucedanoides*. Ril. 31: *Hieracium glaciale*, *Hieracium piloselloides* e *Parnassia palustris*.

Specie sporadiche di Tab. 4 - Ril. 64: *Agrostis tenuis*, *Alchemilla vulgaris*, *Arabis ciliata*, *Botrychium lunaria*, *Carex ericetorum*, *Chenopodium bonus-henricus*, *Gentiana utriculosa*, *Hieracium glaciale*, *Knautia longifolia*, *Koeleria pyramidata*, *Leontopodium alpinum*, *Myosotis alpestris*, *Phleum alpinum*, *Primula farinosa*, *Silene nutans* e *Veronica fruticulosa*. Ril. 3: *Agrostis tenuis*, *Avenula pubescens*, *Carex ericetorum*, *Larix decidua*, *Primula farinosa*, *Silene nutans* e *Stachys alopecurus*. Ril. 67: *Agrostis alpina*, *Antennaria dioica*, *Buphtalmum salicifolium*, *Carex capillaris*, *Dianthus sylvestris*, *Gymnadenia conopsea*, *Helianthemum nummularium*, *Heracleum sphondylium*, *Koeleria pyramidata*, *Myosotis alpestris* e *Taraxacum alpinum*. Ril. 7: *Avenella flexuosa*, *Clematis alpina*, *Lathyrus pratensis*, *Moneses uniflora* e *Valeriana montana*. Ril. 5: *Cirsium acaule*, *Clematis alpina*, *Potentilla aurea*, *Valeriana montana* e *Viola biflora*. Ril. 1: *Campanula barbata*, *Dicranum scoparium*, *Mnium spinosum* e *Nigritella nigra*. Ril. 56: *Crocus albiflorus*, *Gymnadenia conopsea*, *Hieracium sylvaticum* e *Pinguicula alpina*. Ril. 55: *Cirsium acaule*, *Crocus albiflorus*, *Larix decidua*, *Luzula multiflora*, *Pinus mugo*, *Pyrola rotundifolia* e *Potentilla aurea*.

Specie sporadiche di Tab. 5 - Ril. 78: *Botrychium lunaria* e *Primula farinosa*. Ril. 63: *Botrychium lunaria* e *Primula farinosa*. Ril. 86: *Biscutella laevigata*, *Euphrasia pulchella*, *Pyrola rotundifolia* e *Polygala alpestris*. Ril. 57: *Euphrasia pulchella* e *Tofieldia calyculata*. Ril. 61: *Anthoxanthum alpinum*, *Geranium*

sylvaticum e *Ranunculus montanus*. Ril. 30: *Trifolium pratense* subsp. *nivale*. Ril. 24: *Ranunculus montanus*. Ril. 54: *Cirsium acaule*, *Leontodon hispidus*, *Leucanthemum adustum* e *Potentilla crantzii*. Ril. 68: *Buphtalmum salicifolium*, *Campanula scheuchzeri*, *Carlina acaulis*, *Dianthus sylvestris*, *Gentianella engadinensis*, *Hieracium bifidum*, *Petasites albus*, e *Potentilla nitida*. Ril. 77: *Phyteuma orbiculare* e *Scabiosa lucida*. Ril. 8: *Galium pumilum*, *Leontodon hispidus*, *Luzula sieberi* subsp. *sieberi*, *Phyteuma orbiculare*, *Potentilla crantzii*, *Scabiosa lucida*, e *Valeriana montana*. Ril. 76: *Anthoxanthum alpinum*, *Biscutella laevigata*, *Briza media*, *Campanula scheuchzeri*, *Carex flacca*, *Cirsium acaule*, *Gymnadenia conopsea*, *Horminum pyrenaicum*, *Leucanthemum adustum*, *Lotus alpinus*, *Luzula multiflora*, *Parnassia palustris* e *Trifolium pratense* subsp. *nivale*. Ril. 10: *Carex ornithopoda*, *Helianthemum nummularium*, *Horminum pyrenaicum*, *Luzula luzulina* e *Silene vulgaris*.

Specie sporadiche di Tah. 6 - Ril. 6: *Carex sempervirens*, *Anthoxanthum alpinum*, *Leontodon hispidus*, *Melica nutans* e *Polygonum viviparum*. Ril. 9: *Hieracium sylvaticum*, *Horminum pyrenaicum*, *Polygala alpestris* e *Ranunculus acris*. Ril. 2: *Dactylis glomerata*.

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BIODIVERSITY OF EPIPHYTIC LICHENS AND AIR QUALITY IN THE PROVINCE OF GORIZIA (NE ITALY)

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Abstract: In the last six years the north-east Italian plains were intensively studied for air pollution monitoring with lichen biodiversity measures. To date, more than 3.500 relevés of epiphytic lichen vegetation, based on a standard methodology, were carried out in this area. The present study fills a gap in the exploration of the area, presenting the results relative to the province of Gorizia. The study is based on 335 relevés in 104 stations. The relevés are frequency counts of all lichen species in a sampling grid subdivided into ten rectangles. The sum of the frequencies of all species is the Biodiversity Index of each relevé. The average values of Biodiversity Indices of all relevés taken in the same station is the Index of Atmospheric Purity (IAP) of the station, following a slightly modified approach proposed by Swiss authors. The matrices of the 60 lichen species found in the survey area, and of the relevés/stations were submitted to multivariate analysis (classification and ordination): the results show a predominance of nitrophytic *Xanthorion*-species and a transition from *Parmelion* (prevalent in less anthropized areas) to *Xanthorion* vegetation (favoured by agriculture and more resistant to air pollution). Distribution maps showing presence and abundance of eight selected species (*Candelaria concolor*, *Hyperphyscia adglutinata*, *Parmelia caperata*, *P. sulcata*, *P. subrudecta*, *Physcia adscendens*, *Phaeophyscia orbicularis*, *Xanthoria parietina*) are presented and discussed. Automatic mapping programs were also used to map the distribution of the weighed averages of ecological indices associated to each species in each station of the survey area: a map of eutrophication and one of air humidity are presented and discussed. The IAP map of the province is discussed in terms of air quality levels: no extensive lichen desert does occur and IAP values are generally high, but a few restricted areas show some symptoms of air quality worsening, and would be worthy being monitored by instrumental recording.

Introduction

After the UNESCO Conference of Rio, biodiversity has become a key issue in environmental studies. The widespread use of this term, however, has somehow obscured the efforts for its strict operational definition (see e.g. Whittaker 1972). Today, "biodiversity" is often used in a rather naïf sense, as the mere number of taxa occurring in a given Operational Geographic Unit (OGU, Crovello 1981). The quantification of "biodiversity", however, is a much more complex matter. Firstly, it implies a quantitative comparison with OGUs of more or less equal size and similar ecological conditions, a problem which is far from being solved (see e.g. Nimis 1996). Furthermore, biodiversity is also affected by the common/rareness of the species: two OGUs with the same number of taxa obviously differ in terms of biodiversity if in one case the taxa are common and widespread, in the other narrow-ranging and more or less "endemic". Thus, the general distribution of the taxa should also be taken

into account, which is not an easy task. Finally, the relative abundance of each species should be considered as well: two OGUs might have the same number of species, but their abundance-frequencies could be very different, and this will affect what we intuitively understand as "biodiversity" of those OGUs.

That lichen "biodiversity" is a good indicator of air pollution is not a recent discovery. The first observations of an impoverishment of lichen floras in polluted environments date back to the half of the previous century (e.g. Nylander 1866). However, it is only after the second World War, especially thanks to the efforts of several west-European and North American authors, that this simple observation was transformed into a scientifically measurable fact, with the formulation of several formulas for calculating Indices of Air Purity (IAP) from lichen data, which led to the present, so widespread and successful use of lichens as bioindicators of air

pollution (see e.g. De Sloover 1964, Le Blanc & De Sloover 1970, Hawksworth & Rose 1970, Le Blanc 1971, Ferry *et al.* 1973). One of the authors (P.L.N.) was originally skeptical about many of the IAP indices proposed in the literature, since most of them implied a more or less subjective "quantification" of the "sensitivity value" of each species. His objection was that such "sensitivity values", besides methodological problems concerning their estimate, could well change, for each species, with changing environmental conditions, and that this was very difficult to quantify from an operational point of view (Nimis 1985, 1986). A more convincing approach came from a team of Swiss lichenologists, led by K. Ammann, who tested 20 different IAP formulas in Biel-Bienne, a Swiss town, against direct pollution data (Herzig *et al.* 1985, Liebendörfer *et al.* 1988). These authors surprisingly found that the most predictive measurement was the simplest one: the sum of frequencies of all epiphytic lichen species present on the trunk in a series of quadrants within a sampling grid. This is, in spite of its apparent simplicity, a sophisticated biodiversity index, since such a kind of "IAP" value depends, if the sampling area is constant, on two factors: a) the number of species, b) their frequency-cover. These parameters, in the case of lichens, are long-known to be affected by air pollution. The Swiss approach is superior also at a methodological level, as it does not require any assumption about the "sensitivity" to air pollution of every individual species. These are the reasons why, with some modifications in the sampling strategy (Nimis *et al.* 1990), the "Swiss method" was adopted by the senior author in several case-studies (see e.g. Castello *et al.* 1995), until, in the course of the last decade, it has become almost routine throughout Italy. The main modification adopted by Nimis *et al.* (1990) concerns the size of the sampling area: in the original approach this area is variable, since the sampling grid is adapted to fit half the circumference of each phorophyte. Nimis *et al.* (1990) adopted instead a grid of a fixed size; the main reason for this choice is that frequency counts within grids of different sizes are difficult to interpret in terms of biodiversity/area estimates.

Northeastern Italy, comprising the administrative regions of Veneto and Friuli-Venezia Giulia, is a vast area of 26.217 km², more or less corresponding to the ancient Republic of Venezia. After the pioneering studies of Nimis on the lichen vegetation of Trieste (Nimis 1985), Udine (Nimis 1986), and the northern part of the province of Vicenza (Nimis *et al.* 1989), this area has become the object of

intensive surveys of epiphytic lichen biodiversity related to air pollution monitoring, all based on the same sampling strategy, derived from the "Swiss Method", and hence comparable. The entire Region of Veneto was surveyed in 1991 (Nimis *et al.* 1991), with 662 sampling stations and 2.425 relevés, and re-monitored in 1996 in 180 stations for a total of 750 relevés. The province of Trieste was studied, with the same methodology, in 1993 (Castello *et al.* 1995) on 80 stations for a total of 320 relevés, and the results relative to the province of Gorizia, with 104 stations and 335 relevés are presented here. On the whole, until now 3.495 standard relevés of epiphytic lichen biodiversity were carried out in this area from 1989 to 1995, and c. 70% of the area has been surveyed so far. The aim of this paper is to contribute further to this project, by presenting the data relative to the province of Gorizia, immediately joining that of Trieste (Castello *et al.* 1995) to the west, including 104 stations and 335 relevés. When the provinces of Udine and Pordenone will be completed, this will be one of the largest areas in the world where biodiversity assessments of epiphytic lichens related to air pollution will be carried out with such a high sampling intensity, and with the same methodology.

Survey area

The survey area, which corresponds to the administrative limits of the province of Gorizia, in northeastern Italy, has a surface of 473 km², and, geomorphologically, can be subdivided into four main districts (Martinis 1971, Valussi 1971, Cucchi 1984, see Fig. 1a):

1) Collio district - An area of low eocenic hills substratum (max. elevation 275 m), with Flysch as the main lithological, which is the southernmost portion of the Julian pre-Alps. Most of the natural potential vegetation, constituted by acidophytic oak woods, has been largely substituted by cultivations, especially vineyards. However, due to the hilly nature of the landscape, these vineyards are managed in a rather environment-friendly way, without much use of fertilizers and fungicides. In this district there are neither large urban agglomerations, nor important industrial areas.

2) Isonzo Plain district - A lowland area following the course of the Isonzo river, that can be considered as the easternmost part of the Friulian Plain. The main substratum is calcareous alluvial gravel; the albeit poor soils, however, are exploited for agricultural purposes, which has resulted in the

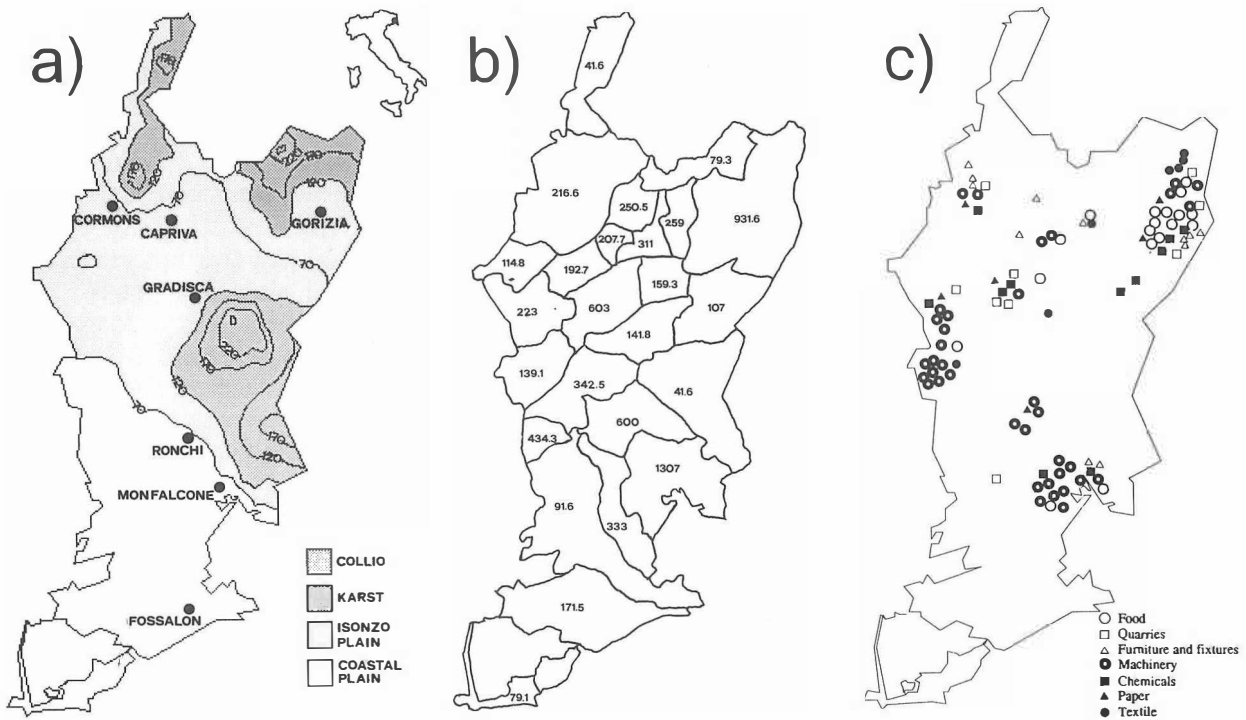


Fig. 1 - Subdivision of the province of Gorizia into four main geomorphological districts: A: Collio, B: Isonzo Plain, C: Karst, D: Coastal Plain; the altitude is indicated by isolines, the main urban centres by circles (Fig. a). Population density (inhabitants/km²) in the municipalities of the province of Gorizia (Fig. b). Location of the main industrial activities in the province of Gorizia (Fig. c).

destruction of most riverine woodlands. This district hosts three important urban centers: Gorizia (38.000 inhabitants), Cormons (7.500 inhabitants) and Gradisca (6.500 inhabitants).

3) Karst district - This is a low calcareous plateau (average elevation: 100 m, max. 275 m) which, to the south-east, borders with the Trieste Karst. Human impact is relatively weak: natural or semi-natural vegetation is prevailing, with open submediterranean woodlands dominated by *Ostrya carpinifolia*, *Fraxinus ornus* and, in more mature stages, by *Quercus pubescens*.

4) Coastal Plain district - This district occupies the southern part of the province, and mainly has fine-textured, clay-silty soils. Most of the area, formerly occupied, especially in its southwestern portion, by swamps, has been reclaimed in the pre- and post-war periods, and natural woodlands have been entirely substituted by intensive cultivations. This district hosts two important centres, which almost constitute a single urban agglomeration, Ronchi dei Legionari (ca. 10.000 inhabitants) and Monfalcone (26.800 inhabitants), and an important industrial area, located east of Monfalcone.

Population density and the location of the main industries are shown in Fig. 1(b-c). The population is concentrated in the municipalities of Gorizia,

Gradisca, Ronchi and Monfalcone. The industries are clustered into four main agglomerations, near Monfalcone, Gorizia, Villesse and Romans. The Collio and Karst districts, and the western part of the Coastal Plain are mostly devoid of important industrial activities.

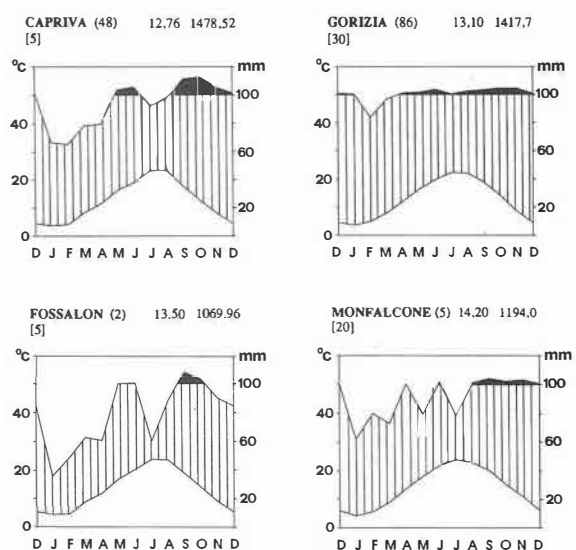


Fig. 2 - Climatic diagrams (according to Walter & Lieth 1960) of four selected localities: Capriva, Gorizia, Fossalon, Monfalcone.

The climate of the province is conditioned by its morphology: the area is under the mild effects of the sea, and relatively protected by the cold northeastern winds ("Bora"), generally blowing during winter. The climatic diagrams of four selected localities are reported in Fig. 2: average yearly temperature tends to decrease from 14.2 °C at Monfalcone in the south to 12.8 °C at Capriva in the north. Yearly precipitations are relatively high, from 1.000 mm near the coast to more than 1.400 mm in the north, precipitation being most abundant in late spring and early autumn, although there is no dry period in the summer season. Average frequency and speed of the winds for two selected stations are shown in Fig. 3(a,b) (Capriva) and Fig. 3(b,c) (Fossalòn): the prevailing winds are from the north (north-east) and from the east. The influence of the strong, dry, cold Bora-wind during winter is more evident in the southern part of the province (Coastal Plain district). On the whole, however, the survey area enjoys a mild, typically submediterranean climate, without summer drought and extreme climatic conditions during winter.

Data and Methods

Relevés of epiphytic lichen vegetation were carried out on the following phorophytes: *Tilia* spp.

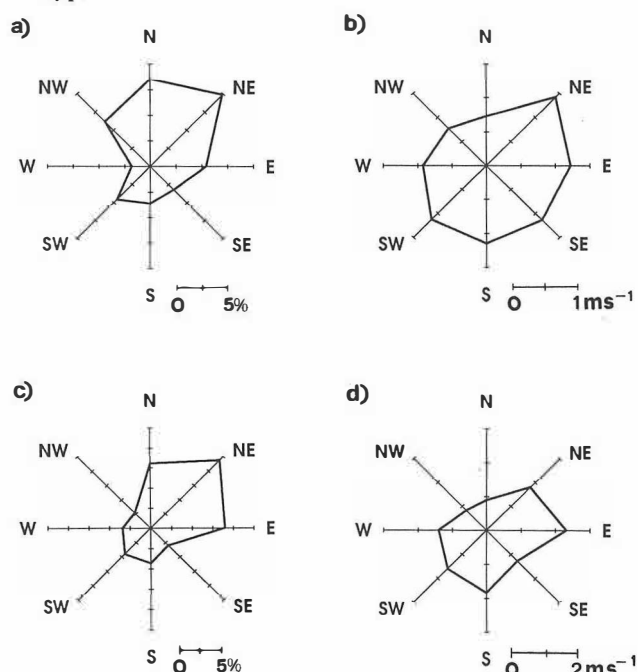


Fig. 3 - Average yearly frequency and average yearly speed of the main winds at the meteorological station of Capriva (a, b) and Fossalòn (c, d).

(254 relevés), *Quercus* spp. (38 relevés) and *Populus* spp. (43 relevés). *Tilia* was selected as the preferential phorophyte because of its mesotrophic bark with subacid pH (Barkman 1958), and as it was largely used in most of the analogous surveys carried out in NE Italy (e.g. Nimis *et al.* 1989, 1991, Castello *et al.* 1995). The bark of *Quercus*, although tendentially more acid and more oligotrophic, is relatively similar to that of *Tilia* (se e.g. Loppi & Putortù 1995). On the contrary, *Populus* has a subneutral, eutrophic bark, and the data collected on *Populus* cannot be directly compared with those deriving from the other phorophytes. For this reason, an *ad hoc* study was carried out in several stations where *Tilia* and *Populus* were co-existing. The results of this study, which will be published in a forthcoming paper, permitted to obtain a transformation factor for IAP values calculated on *Populus*.

Relevés were carried out on phorophytes satisfying the following parameters: a) inclination of the trunk not higher than 10°, b) circumference larger than 70 cm, c) absence of evident factors of disturbance. A total of 335 relevés was carried out in 104 sampling stations (c. three relevés per station), reported in Fig. 4. The selection of the stations occurred in two steps: a) random choice of ca. 2/3 of the total number of stations, b) preliminary elaboration of the IAP data relative to those stations, c) selective choice of further stations in areas which proved to be not sufficiently covered by the first set of stations, especially in those with a strong spatial variation of IAP values.

Relevés were taken using a sampling grid of 30 x 50 cm, subdivided into 10 rectangles of 10 x 15 cm each. The centre of the grid was positioned in the part of the bole with the highest lichen coverage, at an height of c. 150 cm. A relevé listed all species found within the grid, with the number of grid units in which every species occurred (frequency value) being computed. The sum of the frequency values of all species is the Lichen Biodiversity Index (LBI) of the relevé. The average of LBI values of the relevés of each station is the LBI of the station (henceforth called IAP index).

The values of the ecological indices proposed by Wirth (1980), transformed into an ordinal scale as suggested by Nimis *et al.* (1987), were associated to each species. The weighed averages of these values per relevé, using presence-absence data, were used to characterize the ecology of the clusters of relevés, and, in the case of stations, for mapping purposes (see results section).

The matrix of species and relevés (the latter reduced to 317, because some of them were within lichen desert areas), and that of species and stations, were submitted to numerical classification, in order to detect clusters of species with similar ecology, and clusters of floristically similar relevés/stations. The dendrogram of relevés was obtained on the basis of the Coefficient of Correlation, that of stations on the basis of Euclidean Distance. Minimum Variance was used as a clustering algorithm. The matrix of species and relevés was further submitted to Reciprocal Ordering ordination, to detect possible ecological gradients, and to extract a limited number of indicator species. Multivariate analyses were carried out with MULVA (Wildi & Orlóci 1984).

All isoporic maps were obtained by computerized automatic mapping, using the package SURFER (Golden Software Inc., Golden, Colorado). These programs are based on a grid, whose mesh size depends on the minimum and maximum values of the input data. The interpolation method utilized to create a regular grid starting from points (stations) is based on the values of the ten nearest stations. The influence of a given station on a grid point is inversely proportional to their distance.

Nomenclature follows Nimis (1993).

Results

Two data sets are available for this study: the frequencies of all lichen species in the 335 relevés (plus the biodiversity indices of the relevés), and the average frequencies of all species in the 104 stations (plus the IAP indices of the stations). The former have been analyzed in order to give information on the lichen vegetation of the survey area, the latter were used for mapping purposes.

Analysis of the relevés

Altogether, 60 species of lichens were found in the survey area (see Tab. 1), a number which is in good agreement with analogous studies in areas of comparable size and ecological conditions (Nimis *et al.* 1989). The dendrogram of the 317 relevés is schematically shown in Fig. 5, the ordered table is reported in Tab. 1, and the geographic location of the clusters of relevés is shown in Fig. 6 (a,b). In the following, the five main clusters of relevés resulting from the classification will be briefly commented on.

Cluster 1 (71 relevés, 43 species): this cluster is characterized by the highest frequency and / or abun-

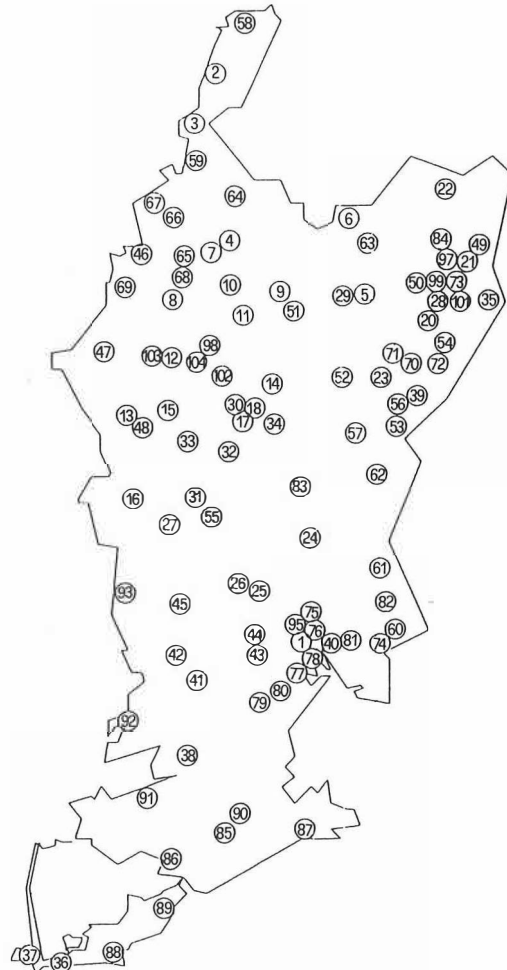


Fig. 4 - Location of the 104 sampling stations in the survey area. The stations are numbered as in Tab. 2.

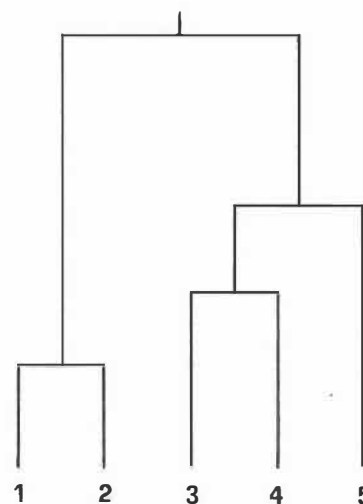


Fig. 5 - Dendrogram of the relevés, based on the data of Tab.1, showing only the five main clusters (minimum variance clustering, correlation coefficient).

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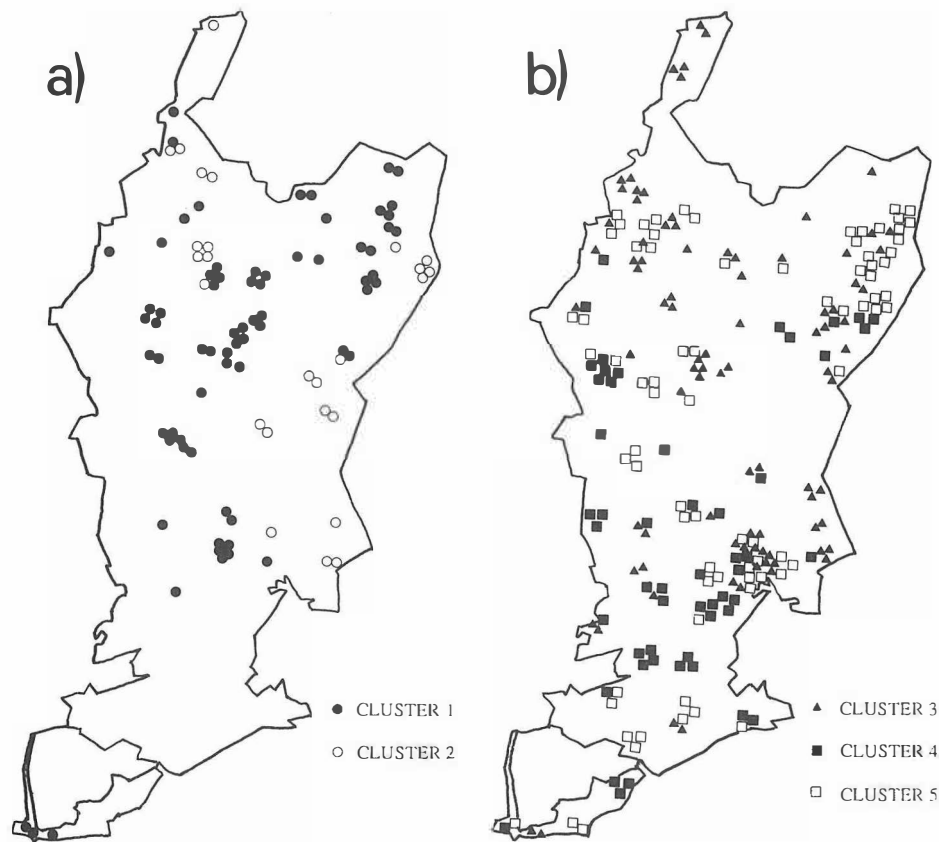


Fig. 6 - Distribution of the 5 clusters of relevés in the survey area:
a) clusters 1-2, b) clusters 3-5.

dance of a set of acidophytic species, such as *Evernia prunastri*, *Hypogymnia tubulosa*, *Hypogymnia physodes*, *Parmelia pastillifera*, *Parmelia saxatilis*, *Parmelia subaurifera*, *Parmelia sulcata*, *Pseudevernia furfuracea*, *Usnea subfloridana*, which are scarcely represented in the other clusters. Lichen Biodiversity Indices are the highest of the whole data set (Fig. 7). Most of the relevés were taken on the northern side of the boles (Fig. 7). This lichen vegetation, which is clearly related to the *Parmelion*-alliance (Barkman 1958, Nimis 1982), is practically absent from the Coastal district (Fig. 6a).

Cluster 2 (26 relevés, 36 species): floristically similar to cluster 1, this cluster differs in the scarcity of several strictly acidophytic species, in the presence of a set of suboceanic species indicative of relatively high air humidity, such as *Lecanora strobilina*, *Normandina pulchella*, *Parmotrema chinense* and *Physcia clementei*, and in a lower frequency of some common nitrophytic lichens such as

Hyperphyscia adglutinata, *Candelaria concolor* and *Candelariella reflexa*, the latter being substituted by the less nitrophytic *Candelariella xanthostigma*. Also in this case LBI values are generally high (Fig. 7). Most of the relevés were taken on the northern part of the boles (Fig. 7). Like in the previous case, also the relevés of cluster 2 are absent from coastal areas (Fig. 6a), their distribution being restricted to the northern and eastern parts of the survey area (Collio and Karst districts).

Cluster 3 (89 relevés, 40 species): this cluster marks an evident vegetational transition: it is characterized, as the following clusters 4 and 5, by the dominance of a few species, most of which are characteristic of *Xanthorion*-vegetation. *Parmelia caperata* and *Parmelia subrudecta*, however, are still present with high frequency here, so that cluster 3 could be considered as transitional between *Parmelion* and *Xanthorion*. On the whole, the floristic composition corresponds well with that of

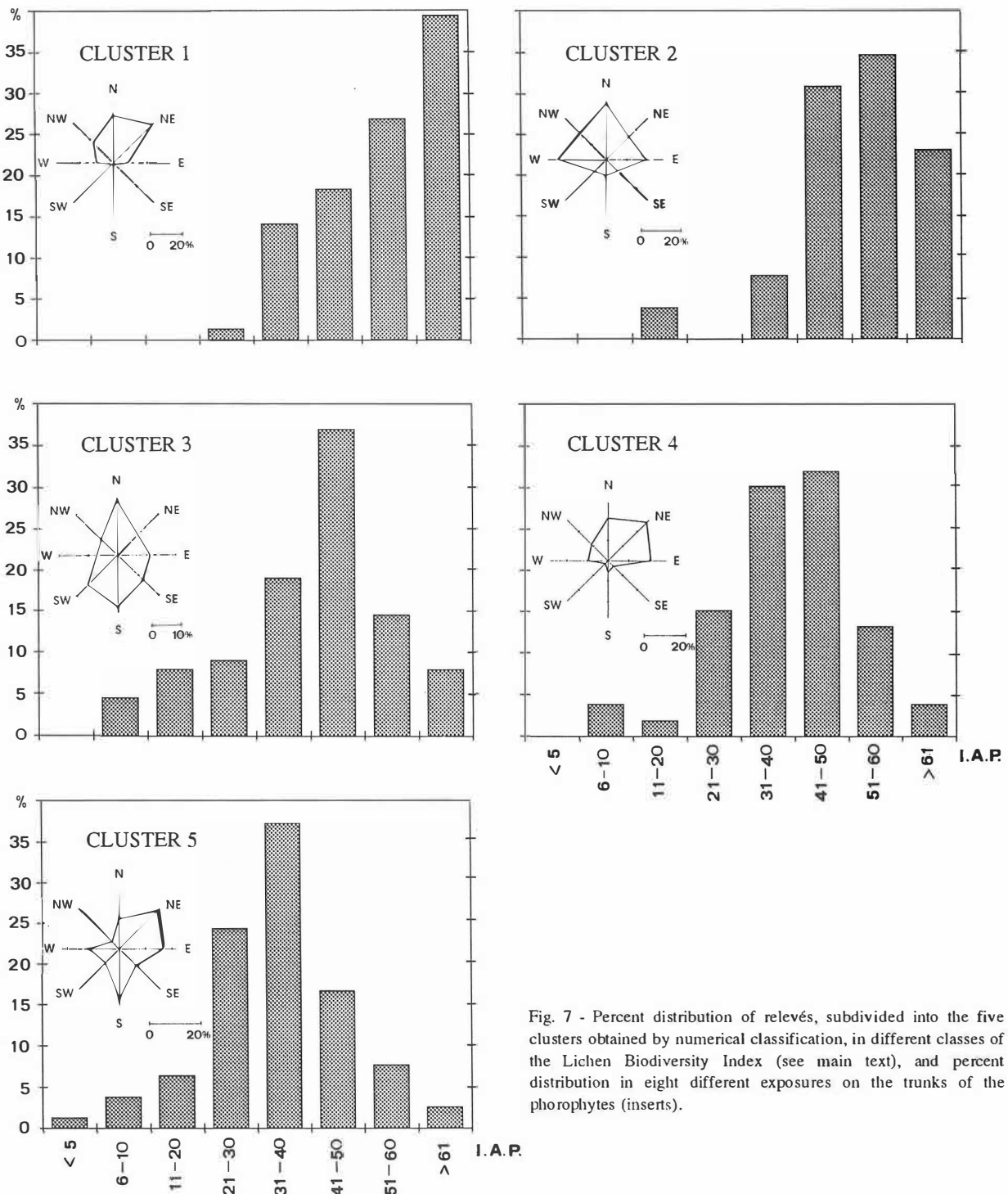


Fig. 7 - Percent distribution of relevés, subdivided into the five clusters obtained by numerical classification, in different classes of the Lichen Biodiversity Index (see main text), and percent distribution in eight different exposures on the trunks of the phorophytes (inserts).

the "*Physcietum elaeinae candelariosum*" described by Nimis & De Faveri (1981), the most common epiphytic synusia on isolated trees in the plains of northeastern Italy (see also Nimis *et al.* 1989, 1991). The LBI values (Fig. 7) tend to be lower than those of the previous clusters. The location of the relevés

around the boles is much less selective, many relevés being located also on the south-exposed sides of the trunks (Fig. 7). Geographically, the relevés are distributed throughout the survey area, with higher frequency in its northern, central and eastern parts (Fig. 6b).

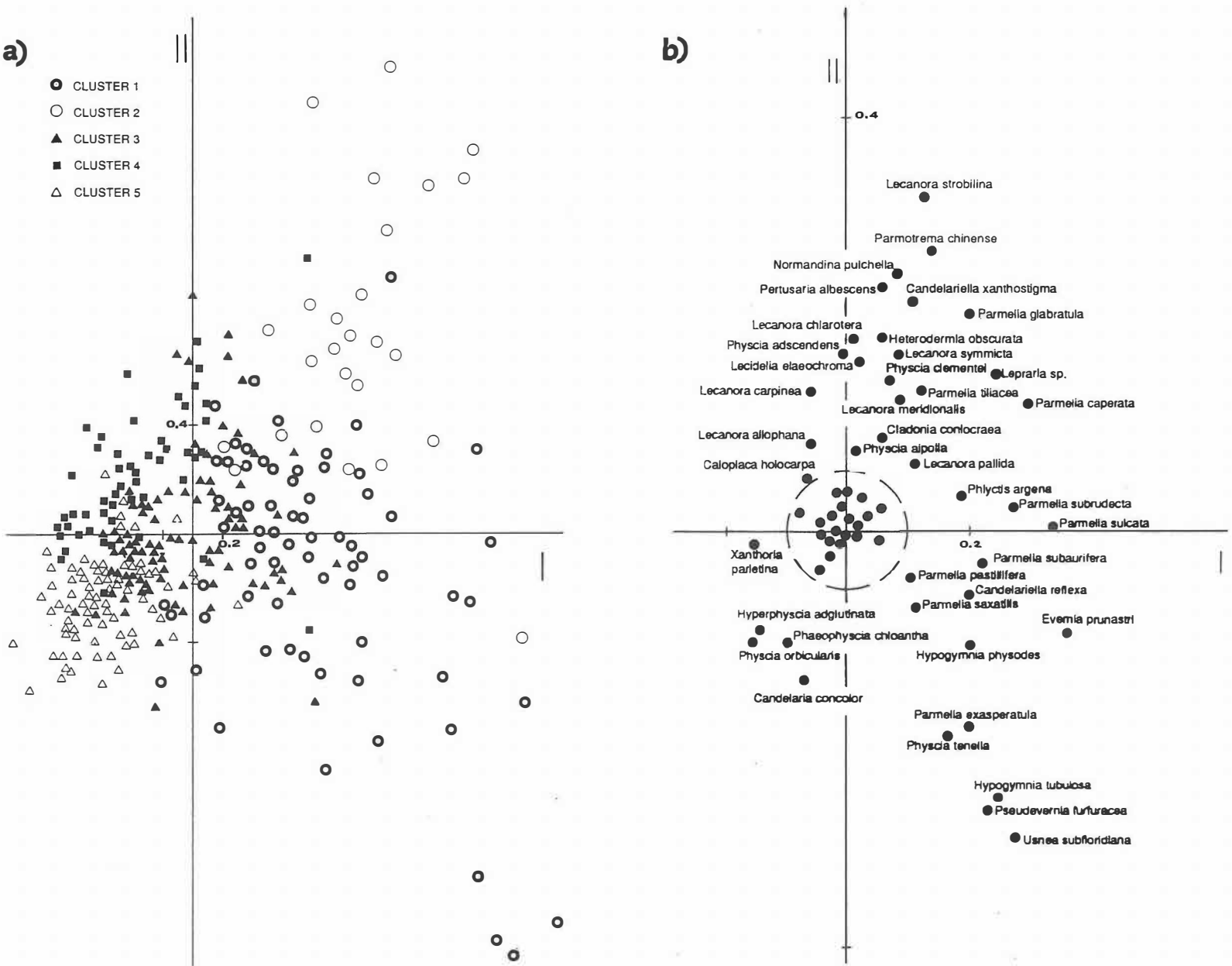


Fig. 8 – Arrangement of relevé (a) and species points (b) in the space defined by the first two axes of the Reciprocal Ordering ordination based on the data of Tab. 1. In Fig. 9a the clusters of relevés are indicated by symbols, as in the legend, in Fig. 9b only the species with high scores on either axis (indicator species) are shown.

Cluster 4 (53 relevés, 35 species): also this cluster has a prevalence of *Xanthorion*-species. It differs from the previous one by the lesser incidence of the nitrophytic species *Candelaria concolor* and *Hyperphyscia adglutinata*, which are substituted by *Phaeophyscia orbicularis* and *Physcia biziana*. On the whole, this is a slightly less nitrophytic variant of cluster 3. Biodiversity tends to be slightly lower than in cluster 3 (Fig. 7), and most relevés are from the northern side of the boles (Fig. 7). The relevés of this cluster are most frequent in the western part of the province, and in the Coastal Plain district (Fig. 6b).

Cluster 5 (78 relevés, 30 species): most species of *Parmelion* are absent from this cluster, which is dominated by a few *Xanthorion*-species, such as *Candelaria concolor*, *Phaeophyscia orbicularis* and *Physcia adscendens*. This is the cluster with the lowest Lichen Biodiversity Indices (Fig. 7). The relevés were taken both at the south and north sides of the boles, with a prevalence of NE and S positions (Fig. 7). The relevés are distributed throughout the survey area, but tend to be concentrated near the largest urban agglomerations and industrial areas (Fig. 6b), which suggests that their floristic composition could be affected by air pollution.

The Reciprocal Ordering ordinations of relevés and species are shown in Fig. 8 (a,b). In the ordination of relevés (Fig. 8a), the first axis clearly separates clusters 1 and 2 (positive scores) from clusters 4 and 5 (negative scores), cluster 3 being intermediate. The second axis mainly separates cluster 1 (negative scores) from cluster 2 (positive scores). This arrangement of points is related to LBI values, which show a clear trend along the first axis (Fig. 9), relevés with positive scores tending to have higher values than those with negative scores. The ordination of the species (Fig. 8b) shows that nitro-, xero- and heliophytic *Xanthorion*-species, which in this part of Europe are generally more tolerant to air pollution (see Nimis 1985, 1986), have negative, or low positive scores on the first axis, while most acidophytic, relatively hygrophytic and less heliophytic, more pollution-sensitive *Parmelion*-species have high positive scores. The arrangement of the relevé points along the first axis of Fig. 8a can be interpreted as a gradient of decreasing nitrophytism and increasing acidophytism, which is paralleled by an increase of the Lichen Biodiversity Index, and by a transition from *Xanthorion*- to *Parmelion*-vegetation. It could be also interpreted as a gradient of decreasing anthropization and decreasing air pollution (see later). The second axis

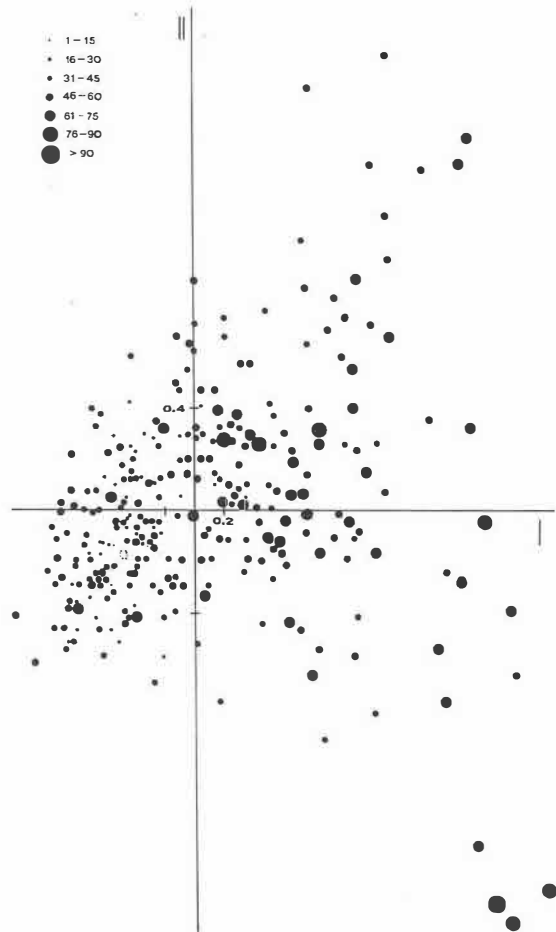


Fig. 9 - Distribution of the Lichen Biodiversity Index in the ordination of relevés of Fig. 8a. The values of the index are represented by dots of increasing size, as in the legend.

mainly accounts for the variation in the two most species-rich clusters (1 and 2), and reflects a gradient in air humidity, limited to clusters 1 and 2: the relevés with positive scores (cluster 2) are those with the highest incidence of relatively more higrophytic lichens, those with negative scores (cluster 1) have the highest incidence of more xerophytic species.

Analysis of the stations

The average frequency data of all species in the 104 stations were also submitted to numerical classification. The dendrogram of the stations is in Fig. 10, and the ordered matrix - in which the species are also arranged according to the results of a numerical classification (not shown) - is reported in Tab. 2. Four main groups of stations are formed (1-4). Group 1 is characterized by a high incidence of *Parmelion*-species. Group 2, while maintaining

Tab. 2 - Average frequency values of the species in the 104 stations, which are numbered as in Fig. 2.

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this feature, also hosts several suboceanic species. Groups 3 and 4 clearly have an higher incidence of *Xanthorion*-species, the latter differing in a lower number of species per station. On the whole, these results are in good agreement with those relative to the analysis of relevés, which means that the vegetational situation within every single station tends to be homogeneous.

The average values of Wirth's ecological indices in the stations were processed by automatic mapping programs to test the use of the indices for mapping some main ecological features of the territory. The geographic distribution of the index of nitrophytism is shown in Fig. 11a: the result is in good agreement with what is known about land use: the most eutrophicated areas are those of the coastal district,

with intensive agriculture, the less eutrophicated those of the Collio and Karst districts, with the lowest impact of agricultural activities. The geographic distribution of the index of hygrophytism is shown in Fig. 11b. This map is almost the opposite of the previous one: most xerophytic species occur in areas with high eutrophication, while most hygrophytic species are more frequent in semi-natural areas with low human impact. This is quite understandable considering that most of the plains are reduced to an "agricultural steppe" while the Collio and Karst districts, besides having a more humid climate, host a much more developed woody vegetation influencing the local microclimatic conditions towards still more humid situations.

The data of Tab. 2 were also used to draw

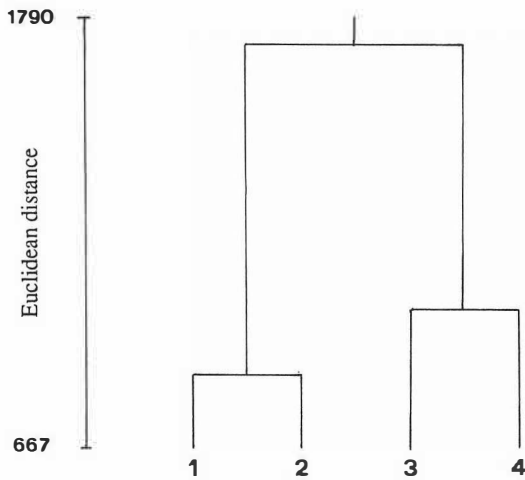


Fig. 10 - Dendrogram of the stations, based on the data of Tab. 3, showing only the four main clusters (minimum variance clustering, euclidean distance).

distribution maps of some of the most common species. A selection is shown in figure 12, reporting, the distribution patterns of *Parmelia caperata*, *P. sulcata*, *P. subrudecta*, *Candelaria concolor*, *Hyperphyscia adglutinata*, *Phaeophyscia orbicularis* and

Xanthoria parietina. These maps are briefly commented on in the following.

The three species of the genus *Parmelia* s.lat. tend to be most frequent in the northern part of the province, being almost absent in the Coastal Plain. In particular, *Parmelia caperata* has its optimum in relatively elevated areas with semi-natural vegetation (open woodlands) such as the Collio and Karst districts, *Parmelia sulcata* is restricted to the northern part of the province, but is able to penetrate within the urban area of Gorizia, probably because of the relatively humid climate and the acidification of bark caused by air pollution; the capability of this species to re-colonize relatively polluted areas was reported for the town of Udine (Nimis 1986), and for the low Venetian Plain (Nimis *et al.* 1991). *Parmelia subrudecta* has a similar distribution pattern, but is able to expand widely into the Isonzo Plain district, which accords well with its ecology: this species is somehow intermediate between *Xanthorion* and *Parmelion*, being most frequent in somehow eutrophicated *Parmelion* stands, and in relatively less eutrophicated *Xanthorion* vegetation, always in situations of low air pollution. The remaining species clearly belong to the *Xanthorion* element

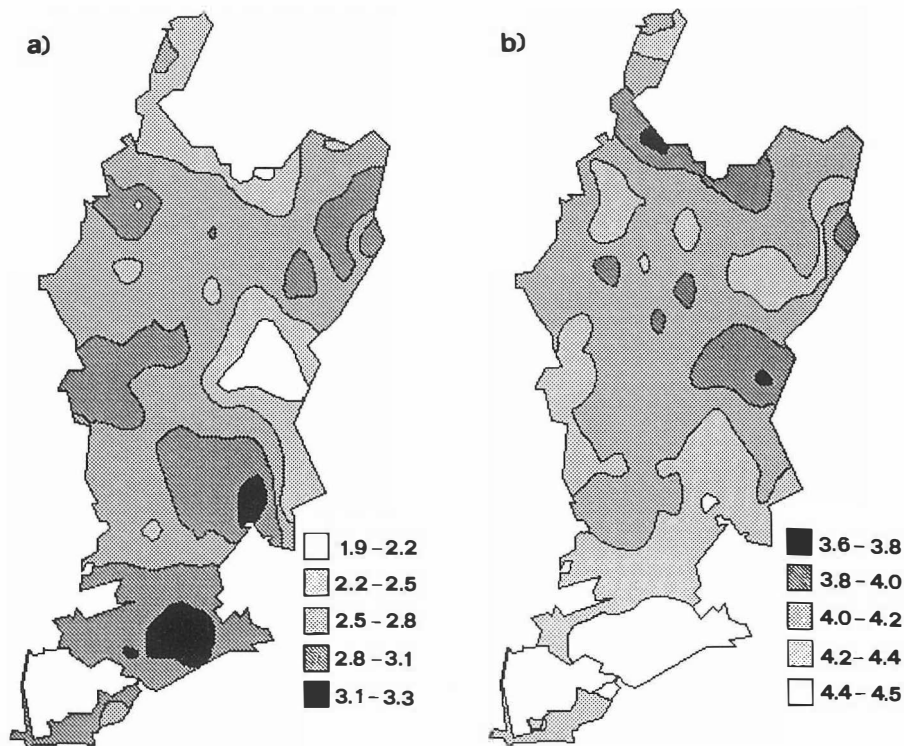


Fig. 11 - Isoporic map showing the distributions of the weighted averages of Wirth's index of nitrophytism (a) and of hygrophytism (b) in the 104 sampling stations.

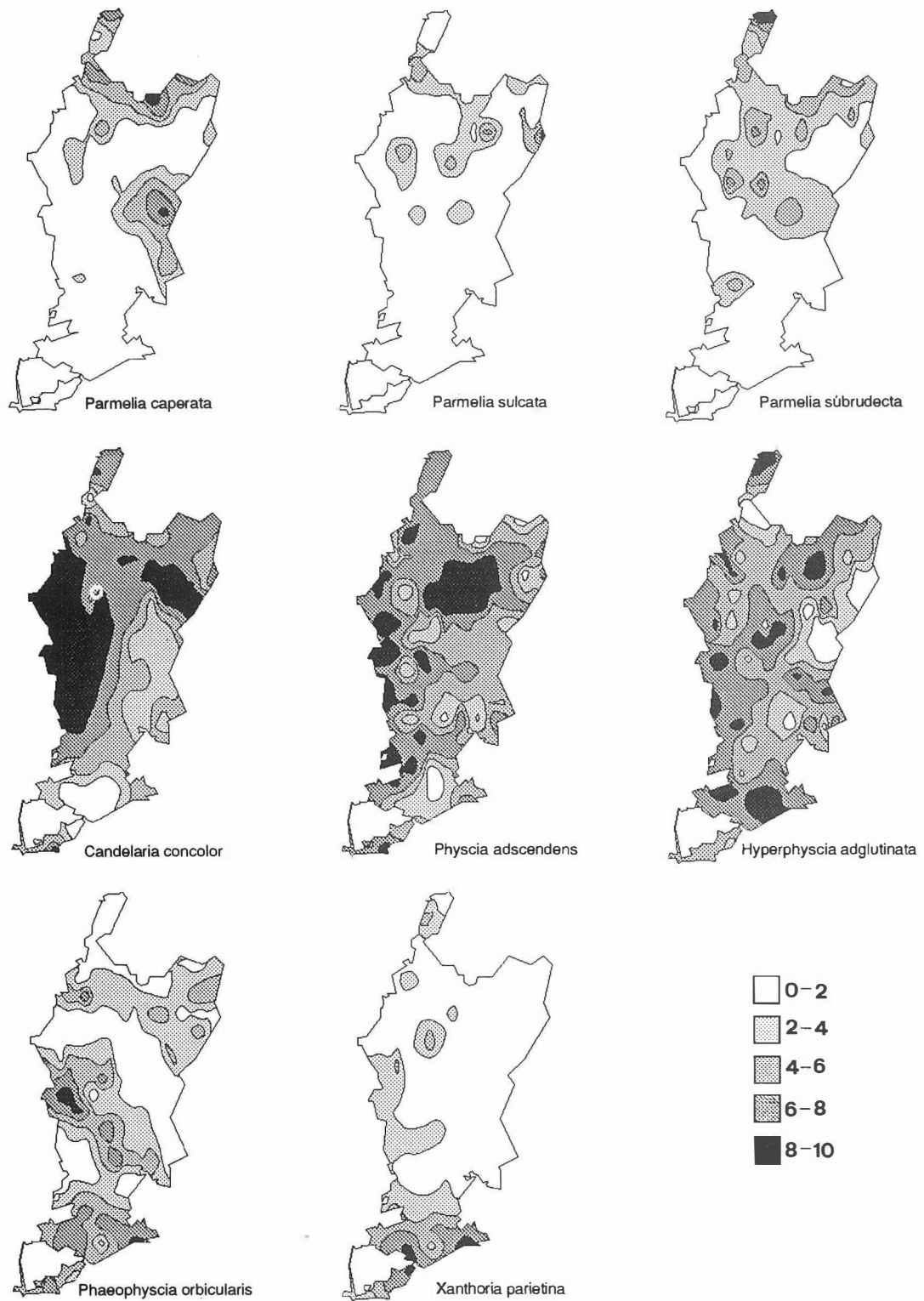


Fig. 12 - Distributions and abundances of eight selected species in the survey area, based on average frequency in the 104 sampling stations.

which is redominant in the survey area: *Candelaria concolor*, *Hyperphyscia adglutinata* and *Physcia adscendens* are the three most common lichens of the province. All of them are typical of eutrophicated situations. This ecological similarity is reflected in a corresponding congruence of distribution pattern: although present throughout the survey area, these species have their optimum outside the Collio and Karst districts, i.e. in the areas which are most heavily exploited by agriculture. The distribution of *Phaeophyscia orbicularis* differs in being mainly western and southern. Finally, *Xanthoria parietina* is clearly most frequent in the Coastal district, perhaps due to the effects of salt spray from the coast towards the interior. Summing up, *Parmelion* and *Xanthorion* species have complementary distributions within the survey area, the former being favoured by more natural conditions, the latter by anthropization, and especially by intensive agriculture (eutrophication).

The map of IAP values relative to the 104 sampling stations is shown in Fig. 13. The survey area has been subdivided into eight belts with different IAP values, using the same intervals adopted by Nimis *et al.* (1991) for the Region of Veneto, and by Castello *et al.* (1995) for the province of Trieste, in order to facilitate the comparison. The highest values were recorded in the Collio district, an area without relevant industries and widespread intensive agriculture; somehow lower values are reached only near the town of Cormons, probably because of urban pollution by domestic heating and local industries (see Figs. 1 b,c). High IAP values also characterize the Isonzo Plain and Karst districts, two areas without relevant concentrations of industries and intensive agriculture. The southern part of the Karst district, however, shows lower IAP values, most probably because of the influence of Monfalcone and its industrial area. This town and its surroundings have the lowest IAP values in the entire province; the negative influence of Monfalcone on IAP values extends towards southwest, in accordance with the direction of the prevailing northeastern winds. Monfalcone is certainly the part of the province with highest air pollution levels, due to three main factors: its relative large population (domestic heating), the presence of a large electric power station, the existence of a large industrial area east of the town. The area surrounding Gorizia also has IAP values lower than 20, although these are limited to a restricted portion of the town. The influence of Gorizia and its industrial area extends some

kilometers southwards, up to the village of Savogna.

When compared with situations known from other parts of northeastern Italy, such as the whole Veneto region (Nimis *et al.* 1991) and the province of Trieste (Castello *et al.* 1995), that of the survey area is, as far as IAP values are concerned, relatively good: no extended lichen desert does occur, the areas with IAP values lower than 20 are very limited, and most of the province has a situation comparable to that of the Dolomites (Nimis *et al.* 1991), or of the Trieste Karst (Castello *et al.* 1995), two areas which are notoriously free from heavy pollution phenomena.

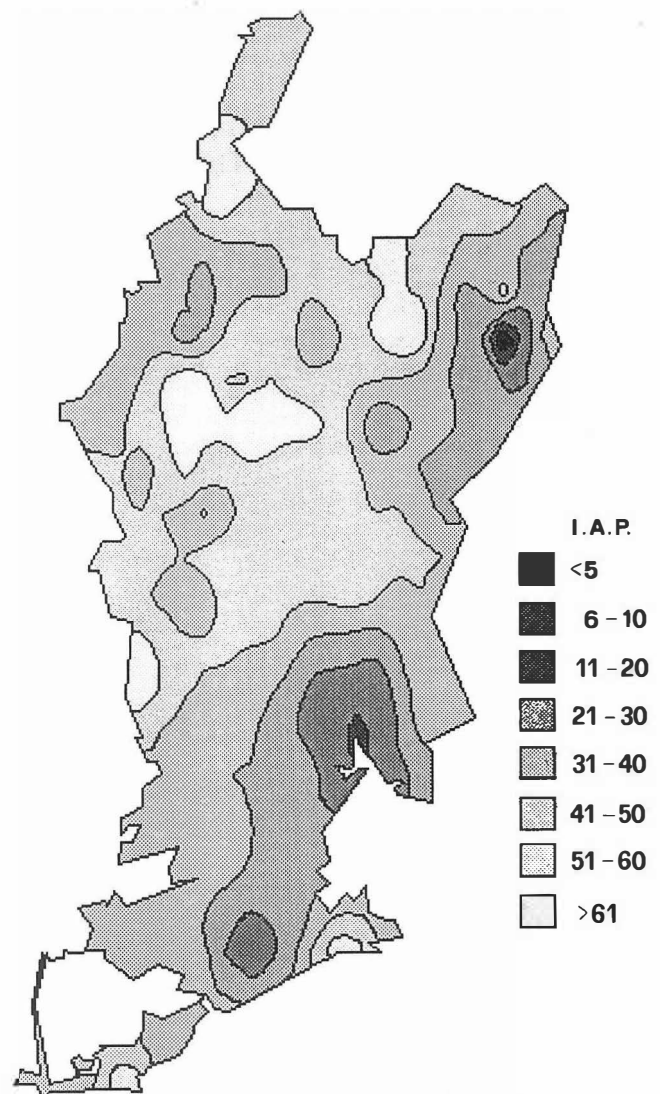


Fig. 13 - Map of the Index of Atmospheric Purity (IAP) in the survey area. The area has been subdivided into 8 belts, as in the legend.

Conclusion

According to Nimis (1990), one of the main reasons for the scarcity of studies relating air pollution with health risk is the scarce availability of reliable pollution data over vast areas, to be used in epidemiological studies. The high costs of recording instruments is a severe constraint on their spatial density. A high spatial density of recording stations, however, is essential for properly monitoring pollution phenomena, which are so variable in space and time. The use of lichen biodiversity values cannot be considered as a substitute of direct pollution recording, also because current legislation, in defining thresholds for given pollutants, refers to actual concentrations in the atmosphere, and these can be measured only by appropriate instruments. Even with lichens, well-known for being among the best bioindicators of air pollution, the interpretation of IAP values in terms of actual air pollution levels is not easy. Biodiversity values, which in our case are limited to number and frequency of species per sampling unit, may depend on several factors other than air pollution, such as climatic conditions, the vegetational situation of the area, and land use. In our study, the climatic situation of the survey area is sufficiently homogeneous as to exclude an important effect on IAP values: some species are clearly more frequent in certain, climatically different parts of the survey area, but the overall IAP map does not show any correlation with climatic parameters. A correlation with land use and natural vegetation, on the contrary, is evident: higher IAP values are reached in the less urbanized and less industrialized areas, i.e. those which host a more natural vegetation. On the contrary, the lowest values are reached near important urban agglomerations and/or industrial areas. This, however, is a further good evidence for the use of lichen biodiversity measurements as indicators of air pollution.

That lichens are sensitive to several different pollutants is now demonstrated by thousands of publications, and nobody would question this well-known fact. The methodology for IAP measurement adopted in this study proved to have a very high predictive value with respect to the sum of several different pollutants in Switzerland (Herzig & Urech 1991). The same type of measurement showed a good correlation with direct data concerning SO₂ concentrations in La Spezia (NW Italy, Nimis *et al.* 1990). Finally, the IAP map of the entire Region of Veneto, obtained through the same method, proved to be extremely well-congruent with a map of death

risk by lung cancer (Nimis & Cislighi, in prep.). Thus, there is unmistakable evidence that lichen biodiversity is strongly related to air quality. The use of these organisms as bioindicators of air pollution has two main advantages: lower costs and, hence, a higher density of sampling stations. In this way the imprecision of the single measurement in terms of air pollution estimate is amply compensated by the high sampling density, which strongly improves the data quality of the maps. The use of lichens as bioindicators can constitute a useful complement to instrumental pollution monitoring in showing the main "high risk areas", and in optimizing the positioning of the few, costly recording instruments on the territory.

In the case of our survey area, compared with the situation prevailing in northeastern Italy, lichens do not suggest the existence of particularly high pollution levels: the only areas which would be worth monitoring in more detail are parts of the Coastal district and the surroundings of the provincial Capital, Gorizia.

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MONITORING OF AIRBORNE METAL POLLUTION BY MOSS BAGS: A METHODOLOGICAL STUDY

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Keywords: metals, monitoring, moss bags, air pollution.

Abstract: The use of moss transplants for monitoring heavy metals deposition is briefly reviewed. The methodological study concerns the effects of different types of pre-treatment on data variability. Epiphytic samples of *Hypnum cupressiforme* were collected from an unpolluted area, treated in different ways, and the resulting bags were exposed in two sites in the province of Trieste (NE Italy) with widely different pollution: a natural woodland far from urban and industrial centers, and a site near an iron smelting industry in the industrial area of Trieste. The content of eight heavy metals (Al, Cd, Cr, Cu, Fe, Mn, Ni, Pb) was measured in 80 moss samples by atomic absorption spectrophotometry. The results, which represent a contribution to the standardization of the moss bags technique, concern: a) variability of metal content in mosses from natural areas: this is influenced by soil contamination; it is advisable to collect epiphytic samples, avoiding those growing near the base of the trunks; b) effects of different washing treatments on metal contents: compared to distilled water, washing with an acid solution is particularly efficient in removing metal ions associated with the cell wall, but produces more variable data; c) uptake capacity of exposed moss bags: this is not influenced by the type of washing, and even short periods of exposure were sufficient to reveal differences in metal depositions between the two sites.

Introduction

In the last thirty years, bryophytes were extensively utilized for assessing the atmospheric deposition of heavy metals and/or radionuclides. Many surveys based on bryophytes were carried out at different geographic scales, and several reviews on this subject are available (e.g. Tyler 1970, 1990, Manning & Feder 1980, Maschke 1981, Martin & Coughtrey 1982, Grodzinska 1982, Rao 1982, Brown 1984, Puckett 1988, Nimis 1990, 1996). The results of intercalibration studies show a good agreement between measurements of heavy metals in mosses and in artificial deposimeters (e.g. Rühling & Tyler 1969, 1970, Tyler 1971, Tyler *et al.* 1983, Pilegaard 1979, Rühling 1985, Rühling *et al.* 1987).

Many bryophytes efficiently entrap dust particles thanks to their peculiar morphology, the absorption of mineral elements being favoured by a large area/volume ratio and by a low surface resistance to ion uptake (Brown 1982, 1984). Bryophytes lack a protective cuticle and thickened epidermal cell walls, and their tissues are easily permeable to water and minerals, including metal ions. Due to the absence of a root system, mineral

nutrition chiefly derives from wet and dry deposition of particles and soluble salts. The substrate is of almost no importance in the nutrition of pleurocarpous mosses and of little importance in most epiphytic mosses (Tamm 1953, Rasmussen & Johnsen 1976, Rasmussen 1978). However, in certain bryophytes, such as terricolous acrocarpous mosses, an uptake of metals from the substratum seems to occur, mainly with rising capillarity water (e.g. Shimwell & Laurie 1972); such forms are less suitable as deposition monitors. The efficiency in particulate trapping mainly depends on features such as the rate of surface hydration, wet bryophytes showing greater efficiency than dry ones. Furthermore, metal levels can be influenced both by canopy cover (Rinne & Barclay-Estrup 1980) and by the sampling position with respect to tree crowns. Metal elements in anthropic or industrialized areas are supposed to be mainly deposited on the surface of bryophytes as particulate matter (Brown 1982, Brown & Bates 1990, Tyler 1990), and according to Brown (1982), bryophytes are morphologically more than chemically suitable to accumulate elements. Their

high water retention capacity, furthermore, reduces the loss of entrapped particles by precipitation (Brown 1982).

Levels of metal accumulation recorded in mosses, usually greatly surpassing those of vascular plants (Czarnowska & Rejement-Grochowska 1974), were also attributed to the cation exchange capacity of moss cell walls (Rühling & Tyler 1970). Ion exchange phenomena were shown to be rapid and passive, and the cation exchange capacity of bryophytes, mainly *Sphagnum*, is closely correlated with the plants content of unesterified polyuronic acid molecules (Clymo 1963, Spearing 1972).

The distribution of elements within the moss tissue is important in understanding tolerance mechanisms, and in planning adequate monitoring strategies. Elements can be found in four possible locations: a) as trapped particulate among the shoots, leaves and rhizoids; b) dissolved in the extracellular solution on the outer surface of the plants, c) within the matrix of the cell walls, bound to the exchange or chelating sites of cell walls (intercellular); d) or as intracellular, soluble or insoluble material in the cytoplasm and vacuoles (Brown 1982, 1984, Brown & Bates 1990). In the first three cases the uptake is passive, while in the last one it is the result of an active biological process, using transport sites through the membrane with various degrees of selectivity.

The use of mosses for monitoring heavy metal deposition is simple and cheap. However, mosses are often absent or scarce in urban areas. For this reason, many authors developed transplants methods. Goodman & Roberts (1971) introduced logs covered with *Hypnum cupressiforme* from an uncontaminated area into an industrial area in Wales, obtaining very high accumulation rates. The transplants died after some weeks but they continued to accumulate metals after death. Pilegaard (1979) found that metal accumulation in exposed samples of *Dicranowesia cirrata* growing on their substratum is linearly correlated with atmospheric deposition. The main problem with transplants is that the samples often cannot survive for long periods in the new habitats, partly due to the adverse local climate, or to severe pollution conditions. Although mosses accumulate heavy metals equally well after death, dead transplants easily start to disintegrate. In areas where the use of indigenous mosses is not possible, exposure of standardized samples, e.g. moss bags, is more advisable.

The moss-bag technique was developed from transplants experiments with *Hypnum cupressiforme*

by Goodman & Roberts (1971). Later authors have usually preferred species of *Sphagnum*, because of its strong resistance to dessication and its high rehydration capacity. Illustrations of the moss bag technique are available in Little & Martin (1974), Ratcliffe (1975), Temple *et al.* (1981), Martin & Coughtrey (1982) and Brown (1984). Moss bags consist of a mesh or grid, generally made of nylon, containing moss shoots whose initial metal content is known; these are often pretreated by distilled water - or acid solution - washing. The bags are suspended in some ways for variable periods in suitable sites before collection and chemical analysis. The main advantages of this technique are the availability of large amounts of material, the uniform and well-defined exposure periods, the possibility of detecting deposition patterns even in "moss desert" areas, thanks to the flexibility of site selection and the high number of sampling sites. Many authors used moss bags to assess deposition patterns of heavy metals linked to industrial emissions (Goodman & Roberts 1971, Little & Martin 1974, Ratcliffe 1975, Gill *et al.* 1975, Cameron & Nickless 1977, Goodman *et al.* 1977, Hynninen 1986); urban emissions (Muskett 1976, Mäkinen 1977), and traffic (Crump & Barlow 1980, Temple *et al.* 1981).

There are variable options in the type of treatment, size and shape of the bags, processing of moss shoots, and habitats in which they are utilized. Some methods used for cleaning bryophyte material and the problems involved are discussed by Brown (1982). A particularly controversial point is still represented by the type of washing: mosses are usually washed in order to remove dust or soil particles and to detect control values of metal contents before exposure. Distilled water removes particles trapped on the surface of the moss, and soluble ions (e.g. Na, Mg, K, Ca, etc) accumulated in the intercellular spaces (Bates 1975, Woollon 1975). Acid-washing causes the realese of ions linked to polyuronic molecules of cell walls, which are substituted by protons (Goodman *et al.* 1977). This kind of treatment kills the moss, and hence the subsequent metal uptake is purely passive, but it should enhance the uptake capacity of exposed samples by increasing metal leaching. Comparisons with material lacking the high cation exchange capacity of mosses, as cotton wool, showed that this is much less efficient (Roberts 1972, Mäkinen 1977, Keller 1974). Many authors recommend exposure for relatively short periods, to avoid saturation of exchange sites, but most reports show a linear increase of metal concentrations with time during

constant emission periods. Many authors carried out methodological studies in order to propose a standard method, which is necessary if absolute deposition rates are to be obtained (e.g. Little & Martin 1974, Gailey & Lloyd 1986a, b, c, Kelly *et al.* 1987). No general agreement, however, has been hitherto reached concerning pretreatment of material, density, size, way and time of exposure of the samples.

The previously discussed methodological problems, and the recent, increasing interest in moss methods by Italian authors, led us to undertake a methodological study dealing with the moss bag technique. The present paper particularly concerns three main aspects of data variability: a) variability of metal content in samples collected in natural areas; b) effects of different washing treatments on the metal content of samples, and, c) uptake capacity of exposed moss bags.

Materials and Methods

This study was carried out using *Hypnum cupressiforme* var. *filiforme* Brid., a taxon widespread all over Italy. The epiphytic variety *filiforme* was chosen, as it is less affected by contamination from soil than terricolous or saxicolous varieties. *Hypnum cupressiforme* accumulates most metals from atmospheric deposition, and only a small share is absorbed from tree bark (Rasmussen & Johnsen 1976); Rasmussen (1978) did not find any relationship between metal contents in samples of *H. cupressiforme* and in the bark on which they grew.

The original samples were collected in a dolina near the small village of Basovizza, at ca 400 m in the Karst of Trieste (NE Italy). This area is far from important urban and industrial settlements, and, according to previous bioindication studies, it is unaffected by relevant pollution phenomena (Castello *et al.* 1995). The natural vegetation consists of a mixed deciduous forest dominated by *Quercus cerris*, *Q. petraea* and *Carpinus betulus* (*Asaro-Carpinetum betuli*, *Seslerio-Quercetum petraea*, Poldini 1989).

The variability of metal contents of *H. cupressiforme* carpets growing in the natural site was studied comparing data from individual carpets with those from composite samples, obtained from a mixture of several individual carpets. Individual samples were collected at different heights, from 10 to 120 cm from the ground, on five trunks of *Quercus*, and separately placed in envelopes. For the preparation of composite samples, several moss carpets were collected from different trees, avoiding

the basal parts of trunks, and carefully mixed together. All samples were cleaned in the laboratory from soil particles, without separating the brown, old parts of the stems from the green ones, except for basal, very old and necrotic parts, which were strongly incrustated with soil material. 10 individual samples and 10 composite samples taken from the mixed material were used for measuring metal concentrations. These samples were not submitted to washing treatments.

The study of the effects of different washing types on metal ion content and uptake capacity of mosses was carried out on the mixed material: half of it was submitted to 7 consecutive washings with distilled water, lasting from 5 to 20 minutes; the remaining material was washed in a 1% HNO₃ acid solution (Hynninen 1986) for 1 hour, and then shortly washed for five times with distilled water. After washing, all samples were air-dried. Concentration values for 10 water-washed and 10 acid-washed samples were compared with those of the 10 composite unwashed samples, *i.e.* with the natural metal content of the mosses. These values represent the control data for assessing metal accumulation rates in the remaining material, which was used for the exposed moss-bags.

Moss bags were prepared as follows: the bag consists of a nylon mesh 10 cm x 10 cm wide, with 1 mm mesh (Davies & White 1981), closed by a nylon wire, forming spherical bags 3-4 cm diam. In each bag 400 mg of moss material were placed. This quantity exceeds the 100-200 mg optimal weight proposed by Gailey & Lloyd (1986a), but it assures enough material for following chemical analyses even if parts of the samples are lost during exposure.

The bags were exposed in two sites with different air pollution conditions: the control site was the same where mosses were collected, the other was located in the industrial zone of Trieste, near an iron works and a high-traffic highway. In each site, 10 water-washed and 10 acid-washed bags were suspended, in order to study the effects of washing on uptake efficiency. The bags were attached in couples of one water-washed and one acid-washed sample at the extremities of a 20 cm long wooden stick. The sticks were suspended on the outermost branches of trees, 1.5-2 m from the ground. Care was taken to eliminate all leaves lying above the bags. The exposure time lasted for eight weeks, from June, 21 to August, 13, 1993, a period characterized by the lowest value of seasonal precipitations. According to Gailey & Lloyd (1986b) an exposure period of 8-9 weeks is the most appropriate to assure high and relatively replicable concentrations for most metals in the samplers.

For the measurement of metal concentrations, ca. 150 mg of each sample were dried with a microwave oven and mineralized in teflon containers at 120° C for 8 hours with concentrated HNO₃ in a pressure decomposition system. The analysis was performed by atomic absorption spectrophotometry, using the graphite furnace for Al, Cd, Pb, the air/acetylen flame for Cu, Cr, Fe, Mn, Ni. To check the analytical method, the NBS Standard Reference Material n. 1572 "Citrus Leaves" was used.

Results

Data variability of metal contents in indigenous moss samples collected in natural sites

One of the variables considered in this study is the height from the ground of samples growing on tree boles. The data obtained from the 10 individual samples taken at heights from 10 to 120 cm on different trees are reported in Tab. 1. Although no strong difference among heights is evident, it is advisable to collect samples at more than 1 m height, as the variability of the data tends to decrease with increasing distance from the ground, probably due to terrigenous contamination of the lower samples.

Tab. 1 reports the metal concentrations of the 10 composite samples taken from the mixed material collected in the natural site. Fig. 1 compares the distribution of metal content values of individual and composite samples. Data from individual samples have a larger spread, but the mean values of the two sets of samples are similar.

The variability of metal concentrations in individual samples is high for monitoring purposes. Our results, however, show that it is possible to characterize a given site by analyzing just a few samples taken from a mixture of several randomly collected moss carpets. The resulting measurements are close to the mean of many measurements from individual samples.

Effects of treatments on variability of metal contents in moss samples

Tab. 2 shows the metal concentration values of 10 water-washed and 10 acid-washed samples. The mean concentration values of all metals in washed samples are lower than those of unwashed samples (Tab. 1), with the exception of a slightly higher mean value of Cu in water-washed samples.

Both types of washing cause the partial removal of metal ions accumulated by mosses, but with

important differences. Acid solution has a remarkable removing efficiency for Cd, Cu and Mn. Water-washed samples have lower mean values of Al, Cr and Fe, and higher values of Cd, Cu, Mn, Ni and Pb (Fig. 2). This difference, statistically tested by the t-test (Tab. 2) is highly significative for Pb, Cu, Fe, Mn and Cd, significative for Ni, and non-significative for Al e Cr; the significance values of Pb and Fe are slightly lower than those of Cd, Cu and Mn. These results can be interpreted on the basis of the different location of metal ions in mosses. Distilled water is known to be less efficient in removing metal ions than acid solution, but the number of repeated treatments has a great importance too. Distilled water shows a higher efficiency in removing Al, Cr, Fe and a lower efficiency for Pb and Ni; these metals are usually linked to atmospheric particles, both from litho- and anthropogenic sources, which are easily removed by repeated washings. The acid solution treatment is very efficient for Cd, Cu and Mn, which are usually associated with the cell wall, and can be easily substituted by protons.

The choice of washing type must be based on the removal efficiency of metal ions and on the resulting variability of the data. The variation coefficient of water-washed samples (Tab. 2) is always lower than that of unwashed samples (with the exception of Pb, Tab. 1), and of acid-washed samples (with the exception of Fe, Tab. 2). Acid-treated samples have a higher variation coefficient for Cd, Cr, Mn and Pb than unwashed ones. It must be however considered that the higher variability in acid-washed samples could be influenced by a higher analytical error: the acid-washed samples show low and even very low concentration values of metals and the analytical error exponentially increases as concentration values decrease.

It can be concluded that although acid washing is more efficient in leaching metals, it is possible to treat moss material with distilled water as well, but several repeated washings are advisable.

Uptake capacity and variability of exposed samples

The uptake capacity of moss bags at different pollution conditions, and the data variability of exposed moss bags were considered in this experiment. Tabs. 3-4 and Fig. 3 show the concentration values of metals recorded in moss bags after exposure in the two sites.

Samples from polluted site (Tab. 4) show remarkably higher contents of all metals than those

Tab. 1 - Metal concentration, mean, standard deviation values (ppm) and coefficient of variation recorded in 10 individual samples and 10 composite samples collected in the control site.

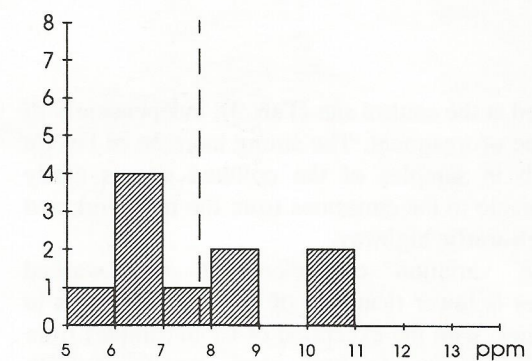
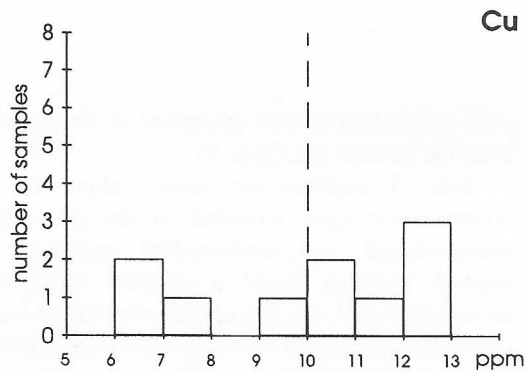
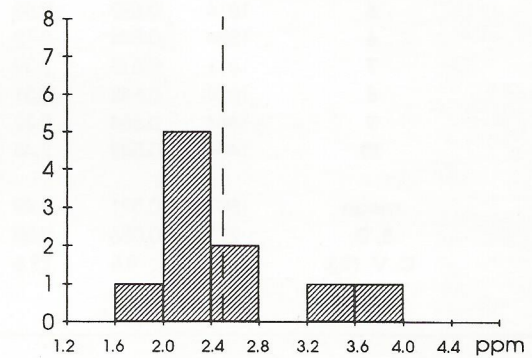
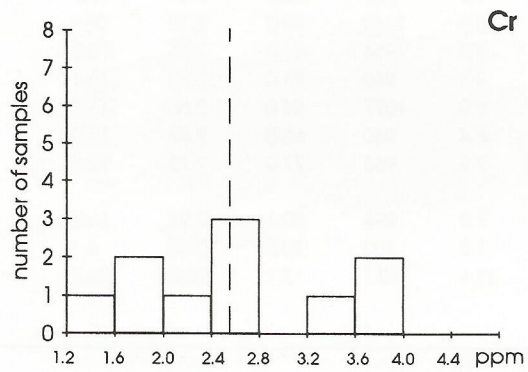
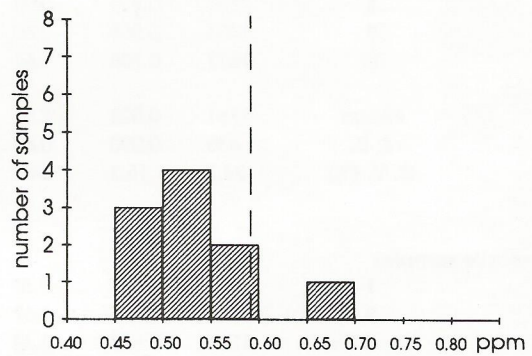
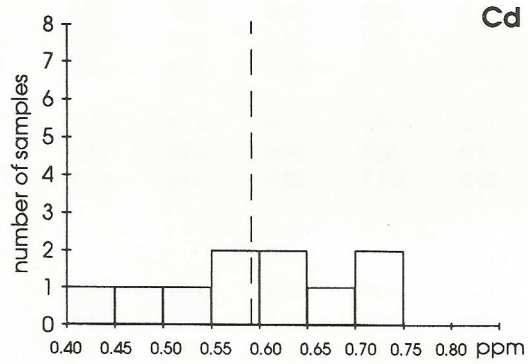
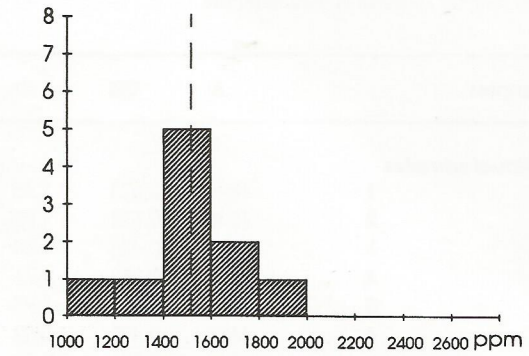
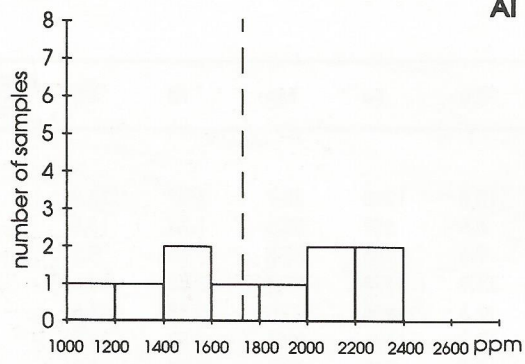
Samples	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Height (cm)
Individual samples									
1	1825	0.628	2.45	10.8	1290	75.7	3.17	33.8	15
2	1235	0.522	1.63	6.5	899	56.6	1.74	13.8	15
3	1113	0.445	1.54	7.6	784	35.5	1.71	5.2	40
4	2176	0.653	2.71	11.9	1398	93.4	3.43	41.6	50
5	2265	0.720	3.97	12.3	1804	121.0	4.82	31.5	60
6	1600	0.492	3.62	13.0	1196	82.4	3.69	31.9	80
7	2069	0.576	2.02	6.2	762	47.1	3.64	4.2	80
8	2320	0.628	3.54	10.0	1654	36.9	3.07	20.6	90
9	1434	0.558	2.60	12.4	950	95.2	3.67	20.5	120
10	1572	0.704	1.63	9.2	885	81.7	2.44	9.5	120
mean	1761	0.593	2.57	10.0	1162	72.6	3.14	21.3	
S. D.	435	0.090	0.89	2.5	368	28.0	0.96	13.1	
C. V. (%)	24.7	15.2	34.7	25.3	31.7	38.5	30.5	61.4	
Composite samples									
1	1690	0.620	3.30	10.2	1137	75.4	4.14	17.9	-
2	1056	0.528	1.67	10.9	821	69.3	2.19	12.3	-
3	1531	0.587	3.72	5.1	926	94.8	4.75	13.9	-
4	1581	0.721	2.23	8.3	997	82.7	2.81	17.9	-
5	1814	0.599	2.45	6.9	1140	98.0	3.10	25.9	-
6	1524	0.544	2.12	6.9	956	69.0	2.35	18.9	-
7	1411	0.618	2.39	8.5	960	83.0	2.65	16.1	-
8	1674	0.588	2.31	6.9	1077	84.0	2.55	18.5	-
9	1384	0.554	2.32	6.4	930	68.0	2.42	14.0	-
10	1401	0.549	2.43	7.9	963	77.0	2.72	12.5	-
mean	1507	0.591	2.49	7.8	991	80.1	2.97	16.8	
S. D.	211	0.056	0.59	1.8	100	10.5	0.83	4.1	
C. V. (%)	24.2	9.5	23.6	22.6	10.1	13.1	28.0	24.2	

exposed at the control site (Tab. 3), independently of the type of treatment. The strong increase of Fe, Cu and Pb in samples of the polluted site is easily attributable to the emissions from the iron work and the high-traffic highway.

The variation coefficient of water-washed samples is lower than that of acid -washed ones in both sites, with the exception of Cr in samples from the control site. In both sites, no significative difference in uptake capacity was found between bags treated with distilled water and those treated

with acid solution, with exception of Fe for samples from the control site (Tab. 5).

Tab. 5 reports the ratio values of mean accumulation rates recorded in the two sites for water-washed and acid-washed samples. Acid-washed samples show a greater efficiency in accumulating Al, Cr, Mn and Pb, the values from the polluted site being from 5.5 to 460 times higher than those of the control site. However, water-washed samples prove to be sensible enough to reveal the different pollution conditions.



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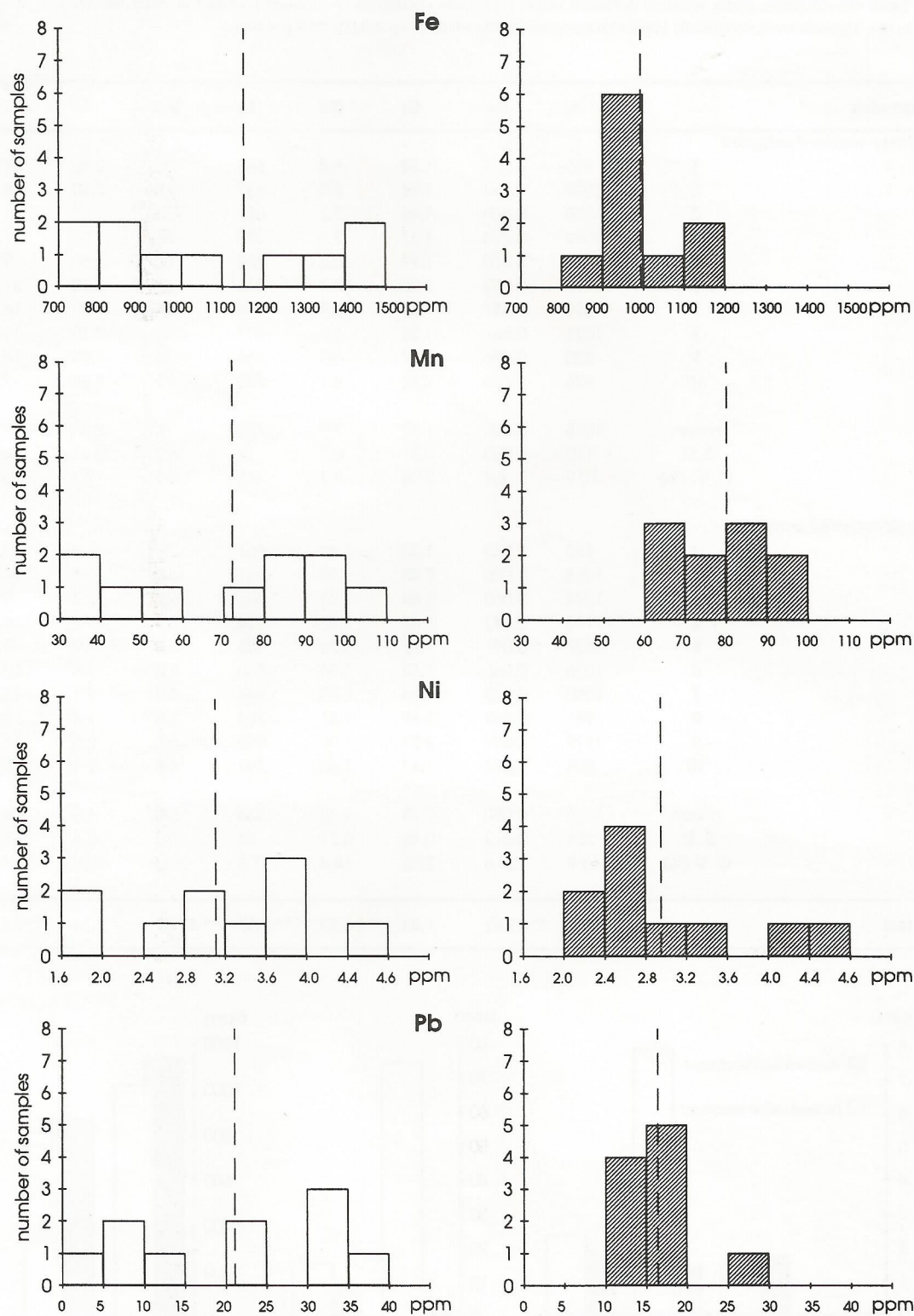


Fig. 1 - Distributions of metal concentration values recorded in 10 individual samples (white bars) and in 10 composite samples (dashed bars). Mean values are reported as dotted lines.

Tab. 2 - Metal concentration, mean, standard deviation values (ppm) and coefficient of variation recorded in water-washed and acid-washed moss samples. The data were statistically compared by means of the t-test. * = $p < 0.05$, ** = $p < 0.01$.

Samples	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb
Water-washed samples								
1	955	0.528	1.39	8.8	683	78	2.60	17.9
2	1029	0.493	1.98	8.4	837	77.8	2.60	18.9
3	1120	0.560	1.46	7.2	858	76	2.60	21.8
4	1078	0.474	1.37	7.9	670	66	1.80	17.8
5	1140	0.570	1.29	8.6	754	82	2.60	9.3
6	1137	0.569	2.00	8.1	870	78	3.10	21.7
7	811	0.437	1.63	6.9	714	67	1.90	14.3
8	1021	0.561	1.24	8.2	803	76	2.10	13.3
9	885	0.496	1.07	6.9	736	73	1.96	16.4
10	975	0.516	1.30	8.1	725	71	2.20	7.2
mean	1015	0.520	1.47	7.9	756	74	2.35	15.9
S. D.	110	0.050	0.31	0.7	72	5.2	0.42	4.9
C. V. (%)	10.9	8.7	21.0	8.7	9.5	6.9	17.7	30.7
Acid-washed samples								
1	782	0.063	1.30	1.40	851	3.9	1.4	10.3
2	1468	0.025	2.45	1.69	957	6.9	2.4	16.9
3	1274	0.060	1.84	1.43	866	6.1	2.3	9.1
4	1115	0.042	1.62	1.48	894	5.3	2	4.5
5	1427	0.041	1.72	1.76	956	4.2	1.9	5.2
6	1076	0.060	1.33	1.54	840	4.6	1.3	10.8
7	1080	0.045	1.94	1.26	844	4.9	2.1	12.2
8	941	0.063	1.19	1.31	764	3.5	1.4	9.5
9	1279	0.047	2.57	1.05	919	5.7	2.5	11.6
10	904	0.054	1.37	1.40	792	4.8	2.1	8.7
mean	1135	0.050	1.73	1.43	868	5.0	1.9	9.9
S. D.	226	0.012	0.48	0.21	64	1.0	0.4	3.5
C. V.(%)	19.9	24.6	27.6	14.4	7.4	20.8	22.4	35.65
t-test								
	1.50	** 41.42	1.44	** 28.69	** 3.37	** 41.79	* 2.13	** 3.14

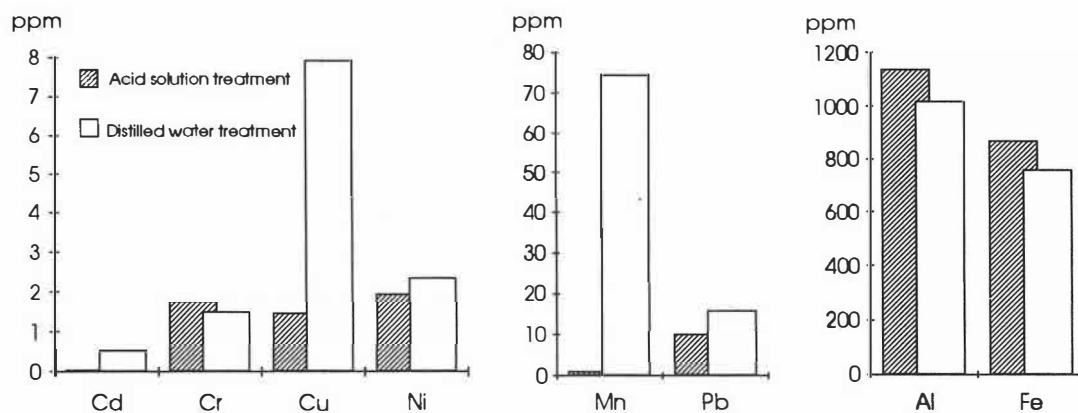


Fig. 2 - Mean metal concentration values recorded in 10 moss samples washed with distilled water (white bars) and in 10 samples treated with acid solution (dashed bars).

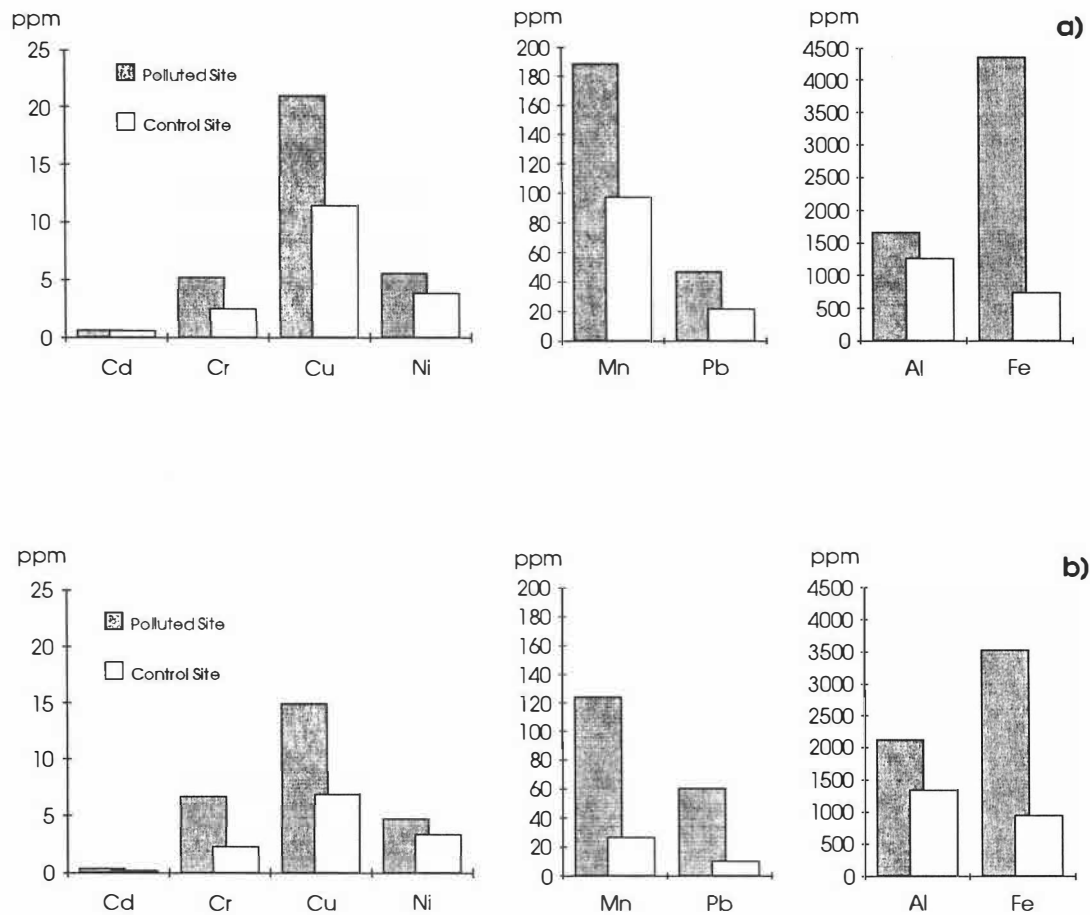


Fig. 3 - Mean metal concentration values recorded in acid-treated (a) and water-treated (b) moss-bags exposed at the control site (white bars) and at the polluted site (dashed bars).

Conclusion

The evaluation of the variability of metal concentrations in individual samples collected in natural areas is important for designing adequate sampling strategies of monitoring studies based on indigenous and on transplanted mosses, or moss bags (see also Gailey & Lloyd 1986a, b, c). Although biological data of metal contamination are usually very variable, our results show that it is possible to characterize a given sampling site by considering just a few moss samples made up of randomly collected and thoroughly mixed material. Attention, however, must be paid to terrigenous contamination.

The cleaning and washing treatments of moss material represent the most important and critical

aspects of the moss bag technique. Distilled water and acid solution act in different ways: the results of this study show that acid washing, a high-impact process that kills the mosses, is highly efficient in removing metal ions usually associated with the cell walls, but produces more variable sets of data. These results differ from those obtained in a similar work by Gailey & Lloyd (1986c), who report a lower variation coefficient in acid-washed samples: the discrepancy could be explained by the efficiency of the acid solution in leaching metals, which could produce variable contents in exchangeable metals, higher degree of difficulty of analytical depending on different acid concentrations or on the different

Tab. 3 - Metal concentration, mean, standard deviation values (ppm) and coefficient of variation recorded in exposed moss-bags at the control site. Samples are distinguished by the different type of treatment.

Samples	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb
<i>Water-washed samples</i>								
1	1772	0.687	6.01	21.6	3766	200	5.22	50.8
2	1762	0.711	6.45	19.1	4090	181	4.65	53.8
3	1690	0.826	6.79	23.2	4315	196	5.21	56.7
4	1674	0.745	3.26	24.0	5528	217	5.72	46.0
5	1496	0.793	5.19	20.6	4727	192	5.61	33.4
6	1723	0.533	5.34	19.0	4385	178	6.89	45.1
7	1540	0.483	3.50	17.9	4003	150	4.9	46.2
8	1608	0.496	5.49	18.0	4129	158	6.4	41.2
9	1724	0.749	5.45	23.0	4597	194	5.5	46.8
10	1646	0.701	4.76	23.4	4036	221	5.5	48.0
mean	1664	0.672	5.22	21.0	4358	189	5.6	46.8
S. D.	92	0.124	1.14	2.4	502	23	0.7	6.5
C.V. (%)	5.5	18.4	21.9	11.3	11.5	12.1	12.1	13.9
<i>Acid-washed samples</i>								
1	2202	0.385	5.73	17.0	4166	138	4.76	65.1
2	2147	0.434	5.37	17.7	4063	131	5.02	61.5
3	2349	0.455	6.59	18.8	4848	161	5.84	81.9
4	2270	0.317	9.29	15.9	3504	133	4.74	55.2
5	2012	0.271	10.2	12.7	3173	122	6.47	58.8
6	2014	0.239	9.73	13.1	3330	108	5.56	49.8
7	2014	0.306	5.27	13.7	2930	119	3.56	53.3
8	1939	0.507	7.91	15.1	3040	111	4.49	49.9
9	2288	0.278	5.69	12.7	3242	109	3.53	64.8
10	1992	0.242	5.35	12.6	2963	108	3.43	64.6
mean	2123	0.343	6.77	14.9	3526	124	4.74	60.5
S. D.	147	0.095	1.76	2.3	632	17	1.03	9.6
C. V. (%)	6.9	27.8	26.1	15.5	17.9	13.8	21.8	15.8

durations of the washing processes; the determination of small quantities of elements must be considered.

Metal ions, and above all those associated on particles, are shown to be highly released by repeated washings with distilled water.

The differences in metal concentrations between samples exposed in non-polluted and polluted sites demonstrate that moss bags are able to detect, even after short exposure periods, different degrees of pollution. The uptake capacity and the efficiency of moss bags seem to be only weakly affected by the type of washing.

The treatment of moss material with several repeated washings with distilled water proved to be

an effective alternative to acid washings in monitoring studies by moss bags.

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Tab. 4 - Metal concentration, mean, standard deviation values (ppm) and coefficient of variation recorded in exposed moss-bags at the polluted site. Samples are distinguished by the different type of treatment.

Samples	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb
Water-washed samples								
1	1350	0.633	3.60	12.6	680	86	3.96	18.0
2	1293	0.639	3.22	13.1	741	120	4.27	28.1
3	1198	0.579	4.52	11.9	724	111	5.00	24.5
4	1234	0.575	3.88	14.7	694	102	4.10	22.1
5	1388	0.509	1.98	12.3	699	93	3.30	25.1
6	1280	0.683	1.61	14.2	727	110	3.10	21.6
7	1220	0.507	2.30	8.3	685	95	3.33	23.8
8	1204	0.599	1.41	7.0	797	74	4.94	16.6
9	1174	0.559	1.45	12.3	826	93	3.81	19.9
10	1235	0.565	1.77	8.8	833	89	2.27	18.9
mean	1258	0.585	2.57	11.5	741	97	3.81	21.9
S. D.	69	0.056	1.13	2.6	58	14	0.84	3.6
C. V. (%)	5.5	9.6	44.1	22.5	7.8	13.9	22.1	16.4
Acid-washed samples								
1	1369	0.144	2.46	4.2	1041	16.2	2.97	9.5
2	1266	0.131	2.33	4.5	909	17.6	2.46	9.1
3	1308	0.199	2.54	8.9	954	25.4	3.81	6.4
4	1361	0.122	2.70	4.2	988	9.4	7.50	13.3
5	1285	0.139	2.28	8.0	1000	28.8	2.85	8.8
6	1279	0.120	2.14	5.2	902	20.8	2.26	11.9
7	1356	0.292	2.39	12.9	968	60.2	3.32	11.1
8	1643	0.345	2.41	9.3	793	24.2	3.40	11.3
9	1273	0.179	2.17	6.5	904	40.9	2.43	7.4
10	1273	0.112	2.17	5.2	1033	21.5	2.87	11.1
mean	1341	0.178	2.36	6.9	949	26.5	3.39	10.0
S. D.	113	0.080	0.18	2.8	75	14.5	1.52	2.1
C. V. (%)	8.4	44.7	7.6	40.8	7.9	54.7	45.0	21.3

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Tab. 5 - Mean concentration values (ppm) of moss samples before and after exposure at the control and at the polluted site. Accumulation data of water- and acid-washed moss bags were calculated as the difference between these mean concentration values, and were statistically compared by means of the t-test. ** = $p < 0.01$. The accumulation efficiency of water- and acid-washed moss bags was estimated by calculating the ratio between accumulation rates at the polluted and at the control site.

Samples	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb
Control Site								
Water-washed samples								
before exposure (A)	1015	0.520	1.47	7.91	756	74.5	2.35	15.9
after exposure (B)	1258	0.585	2.57	11.52	741	97.4	3.81	21.9
accumulation (B-A)	243	0.065	1.10	3.61	-15	22.9	1.46	6.0
S.D.	180	0.106	1.44	3.27	130	18.7	1.26	8.5
Acid-washed samples								
before exposure (A)	1135	0.050	1.73	1.4	868	5.0	1.94	9.9
after exposure (B)	1341	0.178	2.36	6.9	949	26.5	3.39	10.0
accumulation (B-A)	206	0.128	0.63	5.5	81	21.5	1.45	0.1
S.D.	339	0.092	0.66	3.0	139	15.5	1.96	5.7
t-test	0.17	0.85	0.51	0.73	** 7.47	0.10	0.01	1.00
Polluted Site								
Water-washed samples								
before exposure (A)	1015	0.520	1.47	7.9	756	74.5	2.35	15.9
after exposure (C)	1664	0.672	5.22	21.0	4358	188.7	5.56	46.8
accumulation (C-A)	649	0.152	3.75	13.1	3602	114.2	3.21	30.9
S.D.	202	0.174	1.45	3.0	575	27.9	1.09	11.4
Acid-washed samples								
before exposure (A)	1135	0.050	1.73	1.4	868	5.0	1.94	9.9
after exposure (C)	2123	0.343	6.77	14.9	3526	124.0	4.74	60.5
accumulation (C-A)	988	0.293	5.04	13.5	2658	119.0	2.80	50.6
S.D.	373	0.108	2.24	2.5	697	18.1	1.47	13.1
t-test	1.39	1.09	0.84	0.19	1.81	0.25	0.39	1.95
Accumulation ratio								
Water-washed (C-A)/B-A)	2.7	2.4	3.4	3.6	233.9	5.0	2.2	5.2
Acid-washed (C-A)/B-A)	4.8	2.3	8.0	2.5	32.7	5.5	1.9	460.1

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